CHAPTER 8

RECEIVING WATER MODELING

This chapter discusses the use of receiving water modeling to evaluate CSO impacts to receiving waters. It uses the term "modeling" broadly to refer to a range of receiving water simulation techniques. This chapter introduces simplified techniques, such as dilution and decay equations, and more complex computer models, such as QUAL2EU and WASP.

8.1 THE CSO CONTROL POLICY AND RECEIVING WATER MODELING

Under the CSO Control Policy a permittee should develop a long-term control plan (LTCP) that provides for attainment of water quality standards (WQS) using either the demonstration approach or presumption approach. Under the demonstration approach, the permittee documents that the selected CSO control measures will provide for the attainment of WQS, including designated uses in the receiving water. Receiving water modeling may be necessary to characterize the impact of CSOs on receiving water quality and to predict the improvements that would result from different CSO control measures. The presumption approach does not explicitly call for analysis of receiving water impacts.

In many cases, CSOs discharge to receiving waters that are water quality-limited and receive pollutant loadings from other sources, including nonpoint sources and other point sources. The CSO Control Policy states that the permittee should characterize the impacts of the CSOs and other pollution sources on the receiving waters and their designated uses (Section II.C.1). Under the demonstration approach, "[w]here WQS and designated uses are not met in part because of natural background conditions or pollution sources other than CSOs, a total maximum daily load, including a wasteload allocation and a load allocation, or other means should be used to apportion pollutant loads." (Section II.C.4.b)

Established under Section 303(d) of the CWA, the total maximum daily load (TMDL) process assesses point and nonpoint pollution sources that together may contribute to a water body's impairment. This process relies on receiving water models.

An important initial decision-which water quality parameters to model-should be based on data from receiving water monitoring. CSOs affect several receiving water quality parameters. Since the impact on one parameter is frequently much greater than on others, relieving this main impact will likely also relieve the others. For example, if a CSO causes exceedances of bacteria WQS by several hundredfold, as well as moderate dissolved oxygen (DO) depressions, solving the bacterial problem will likely solve the DO problem and so it may be sufficient to monitor bacteria only. Reducing the scope of modeling in this fashion may substantially reduce costs.

8.2 MODEL SELECTION STRATEGY

A receiving water model should be selected according to the following factors:

- The type and physical characteristics of the receiving water body. Rivers, estuaries, coastal areas, and lakes typically require different models.
- The water quality parameters to be modeled. These may include bacteria, DO, suspended solids, toxics, and nutrients. These parameters are affected by different processes (e.g., die-off for bacteria, settling for solids, biodegradation for DO, adsorption for metals) with different time scales (e.g., hours for bacterial die-off, days for biodegradation) and different kinetics. The time scale in turn affects the distance over which the receiving water is modeled (e.g., a few hundred feet for bacteria to a few- miles for DO).
- The number and geographical distribution of CSO outfalls and the need to simulate sources other than CSOs.

This section discusses some important considerations for hydrodynamic and water quality modeling of receiving waters, and how these considerations affect the selection and use of a model.

The purpose of receiving water modeling is primarily to predict receiving water quality under different CSO pollutant loadings and flow conditions in the receiving water. The flow conditions, or hydrodynamics, of the receiving water are an important factor in determining the effects of CSOs on receiving water quality. For simple cases, hydrodynamic conditions can be determined from the receiving water monitoring program; elsewhere a hydrodynamic model may be necessary.

Hydrodynamic and water quality models are either ***steady-state*** or ***transient.*** Steady-state models assume that conditions do not change over time, while transient models can simulate conditions that vary over time. Flexibility exists in the choice of model types; generally, either a steady-state or transient water quality simulation can be done regardless of whether flow conditions are steady-state or transient.

8.2.1 Hydrodynamic Models

A hydrodynamic model provides the flow conditions, characterized by the water depth and velocity, for which receiving water quality must be predicted. The following factors should be considered for different water body types:

- ***Rivers-*** Rivers generally flow in one direction (except for localized eddies or other flow features) and the stream velocity and depth are a function of the flow rate. The flow rate in relatively large rivers may not increase significantly due to wet weather discharges, and a constant flow can be used as a first approximation. This constant flow can be a specified low flow, the flow observed during model calibration surveys, or a flow typical of a season or month. When the increase of river flow is important, it can be estimated by adding together all upstream flow inputs or by doing a transient flow simulation. The degree of refinement required also depends on the time scale of the water quality parameters of interest. For example, assuming a constant river flow may suffice for bioaccumulative toxicants (e.g., pesticides) because long-term exposure is ofimportance. For DO, however, the time variations in river flow rate may be need to be considered.

- ***Estuaries-*** CSO impacts in estuaries are affected by tidal variations of velocity and depth (including reversal of current direction) and by possible salinity stratification. Tidal fluctuations can be assessed by measuring velocity and depth variations over a tide cycle or by using a one- or two-dimensional model. Toxics with relatively small mixing zones can be analyzed using steady currents corresponding to different times during the tidal cycle, but this may require using a computed circulation pattern from a model.

- ***Coastal Areas-*** CSO impacts in coastal areas are also affected by tidal fluctuations. The discussion on estuaries generally applies to coastal areas, but, because the areas are not channelized, two-dimensional or even three-dimensional models may be necessary.

- ***Lakes-*** CSO impacts in lakes are affected by wind and thermal stratification. Wind-driven currents can be monitored directly or simulated using a hydrodynamic model (which may need to cover the entire lake to simulate wind-driven currents properly). Thermal stratification can generally be measured directly.

Because the same basic hydrodynamic equations apply,[1] some of the major models for receiving waters can be used to simulate more than one type of receiving water body. Ultimately, three factors dictate whether a model can be used for a particular hydraulic regime. One factor is whether it provides a one-, two-, or three-dimensional simulation. A second is its ability to handle specific boundary conditions, such as tidal boundaries.

A third factor is whether the model assumes steady-state conditions or allows for time-varying pollutant loading. In general, models that assume steady-state conditions cannot accurately model CSO problems that require analysis of far-field effects. However, in some instances a steady-load model can estimate the maximum potential effect, particularly in systems where the transport of constituents is dominated by the main flow of the water body, rather than local velocity gradients. For example, by assuming a constant source and following the peak discharge plug of water downstream, the steady-load model QUAL2EU can determine the maximum downstream effects of conventional pollutants. The result is a compromise that approximates the expected impact but neglects the moderating effects of longitudinal dispersion. However, QUAL2EU cannot give an accurate estimate of the duration of excursions above WQS.

8.2.2 Receiving Water Quality Models

The frequency and duration of CSOs are important determinants of receiving water impacts and need to be considered in determining appropriate time scales for modeling. CSO loads are

[1] The basic hydrodynamic equations are for momentum and continuity. The momentum equation describes the motion of the receiving water, while the continuity equation is a flow balance relationship (i.e., total inflows to the receiving water less total outflows is equal to the change in receiving water volume).

typically delivered in pulses during storm events. Selection of appropriate time scales for modeling receiving water impacts resulting from a pulsed CSO loading depends upon the time and space scales necessary to evaluate the WQS. If analysis requires determining the concentration of a toxic at the edge of a relatively small mixing zone, a steady-state mixing zone model may be satisfactory. When using a steady-state mixing zone model in this way, the modeler should apply appropriately conservative but characteristic assumptions about instream flows during CSO events. For pollutants such as oxygen demand, which can have impacts lasting several days and extending several miles downstream of the discharge point, it may be warranted to incorporate the pulsed nature of the loading. Assuming a constant loading is much simpler (and less costly) to model; however, it is conservative (i.e., leads to impacts larger than expected). For pollutants such as nutrients where the response time of the receiving water body may be slow, simulating only the average loading rate, usually over a period of days (e.g., 21 days) depending on the nutrient, may suffice.

Receiving water models vary from simple estimations to complex software packages. The choice of model should reflect site conditions. If the pulsed load and receiving water characteristics are adequately represented, simple estimations may be appropriate for the analysis of CSO impacts. To demonstrate compliance with the CWA, the permittee may not need to know precisely where in the receiving water excursions above WQS will occur. Rather, the permittee needs to know the maximum pollutant concentrations and the likelihood that excursions above the WQS can occur at any point within the water body. However, since CSOs to sensitive areas are given a higher priority under the CSO Policy, simulation models for receiving waters with sensitive areas may need to use short time scales (e.g., hourly pollutant loads), and have high resolution (e.g., several hundred yards or less) to specifically assess impacts to sensitive areas.

8.3 AVAILABLE MODELS

Receiving water models cover a wide variety of physical and chemical situations and, like combined sewer system (CSS) models, vary in complexity. EPA has produced guidance on receiving water modeling as part of the Waste Load Allocation (WLA) guidance series. These models, however, tend to concentrate on continuous sources and thus may not be the most suitable for CSOs. Ambrose et al. (1988a) summarizes EPA-supported models, including receiving water models.

This guidance does not provide a complete catalogue of available receiving water models. Rather, it describes simplified techniques and provides a brief overview of relevant receiving water models supported by EPA or other government agencies. In many cases, detailed receiving water simulation may not be necessary. Use of dilution and mixing zone calculations or simulation with simple spreadsheet models may be sufficient to assess the magnitude of potential impacts or evaluate the relative merits of various control options.

Types of Simulation

Water quality parameters can be simulated using either single-event, steady-state modeling or continuous, dynamic modeling. Many systems may find it beneficial to use both types of modeling.

Many of the simpler approaches to receiving water evaluation assume steady flow and steady or gradually varying loading. These assumptions may be appropriate if an order-of-magnitude estimate or an upper bound of the impacts is required. The latter is obtained by using conservative parameters such as peak loading and low current speed. If WQS attainment is predicted under realistic worst-case assumptions, more complex simulations may not be needed.

Due to the random nature of CSOs, the use of dynamic simulation may be preferable to single-event, worst-case, steady-state modeling. Dynamic techniques allow the modeler to derive the fraction of time during which a concentration was exceeded and water quality was impaired. For instance, when using daily simulated results, specific concentrations are first ranked with the corresponding number of occurrences during the simulation period. Frequency distribution plots are then developed and used to determine how often the l-day-acute water quality criteria are likely to be exceeded. The same approach can be used to develop frequency distributions for longer periods such as 4-day or 30-day average concentrations. EPA (1991a) recommends three dynamic modeling techniques: continuous simulation, Monte Carlo simulation, and lognormal probability modeling.

Continuous simulation models solve time-dependent differential equations to simulate flow volume and water quality in receiving waters. These deterministic models incorporate the manner in which flow and toxic pollutant concentrations change over time in a continuous manner rather than relying on simplified terms for rates of change. They use daily effluent flow and concentration data with daily receiving water flow and concentration data to estimate downstream receiving water concentrations. If properly calibrated and verified, a continuous simulation model can predict variable flow and water quality accurately-although at a considerable time and resource expenditure, however.

Monte Carlo simulation is generally used for complex systems that have random components. Input variables are sampled at random from pre-determined probability distributions and used in a toxic fate and transport model. The distribution of output variables from repeated simulations is analyzed statistically to derive a frequency distribution. However, unlike continuous simulation models, the temporal frequency distribution of the output depends on the temporal frequency distribution of the input data. For instance, if the water quality criterion is based on a 4-day average, the input variables must use the probability distributions based on a 4-day average.

Lognormal probability modeling estimates the same output variable probability distributions as continuous and Monte Carlo simulations but with less effort. However, like Monte Carlo simulation, the input must be probability distributions based on input data for the specific temporal frequency distribution desired. The theoretical basis of the technique permits the stochastic nature of the CSO process to be explicitly considered. This method assumes that each of the four variables that affect downstream receiving water quality (rainfall, runoff, event mean concentration of contaminant in the runoff (EMC), and streamflow) can be adequately represented by a lognormal probability distribution. When the EMC is coupled with a lognormal distribution of runoff volume, the distribution of runoff loads can be derived. The storm water load frequency is then coupled with a lognormal distribution of streamflow to derive the probability distribution of in-stream concentrations. The main advantage of lognormal probability modeling is that the probability distributions can be derived using only the median and the coefficient of variation for each input variable.

8.3.1 Model Types

The following sections discuss techniques for simulating different water quality parameters in rivers, lakes and estuaries.

RIVERS

Bacteria and Toxics. Bacteria and toxic contaminants are primarily a concern in the immediate vicinity of CSO outfalls. They are controlled by lateral mixing, advection, and decay processes such as die-off (for bacteria), vaporization (for toxics), and settling and resuspension (for bacteria and toxics). When stream flow is small relative to CSO flow, lateral mixing may occur rapidly and a one-dimensional model may be appropriate. Initial estimates can be made using a steady-state approach that neglects the time-varying nature of the CSO. In this case, concentrations downstream of a CSO are given by:

$$C_x = \frac{Q_u C_u + Q_e C_e}{Q_s} e^{\frac{-KX}{u}}$$

where:[2]

C_x = max pollutant concentration at distance X from the outfall (M/L^3)
C_e = pollutant concentration in effluent (M/L^3)
C_u = pollutant concentration upstream from discharge (M/L^3)
Q_e = effluent flow (L^3/T)
Q_u = stream flow upstream of discharge (L^3/T)
Q_s = stream flow downstream of discharge, $Q_u + Q_e$ (L^3/T)
X = distance from outfall (L)
u = stream flow velocity (L/T)
K = net decay rate (die-off rate for bacteria, settling velocity divided by stream depth for settling, resuspension velocity divided by stream depth for resuspension, vaporization rate divided by stream depth for vaporization) (1/T)
e = 2.71828...

Since bacteria and toxics can settle out of the water column and attach to sediments, sediments can contain significant amounts of these pollutants. Resuspension of sediments and subsequent desorption of bacteria and toxics into the water column can be an important source of receiving water contaminants. Modeling of sediment resuspension requires estimation of

[2] M=unit of mass, L=unit of length, and T=unit of time.

resuspension velocities and knowledge of sediment transport processes. Thomann and Mueller (1987) discusses how to determine the solids balance in a river and estimate sediment resuspension velocities. Modeling of sediment transport is complex and is often done using computer models such as WASP5 and HSPF.

In large rivers, lateral mixing may occur over large distances and bacterial counts or toxics concentrations on the same shore as the discharge can be calculated using the following expression, as a conservative estimate (U.S. EPA, 1991a):

$$C_x \approx \frac{C_e Q_e W}{Q_s \sqrt{\dfrac{\pi D_y X}{u}}}$$

where: D_y = lateral dispersion coefficient (L^2/T)
W = stream width (L)
π = 3.14159...

This equation is conservative because it neglects any discharge-induced mixing. Simulating over the correlated probability distributions of C_e, Q_e, Q_s, and Q_u can provide an estimate of the frequency of WQS exceedances at a specific distance from the outfall. The method requires the estimation of a lateral dispersion coefficient, which can be measured in dye studies or by methods described in Mixing ***in Inland and Coastal Waters*** (Fischer et al., 1979). Fischer's methods calculate the lateral dispersion coefficient D_y as follows:

$$D_y = 0.6\ du^* \pm 50\%$$

where: d = water depth at the specified flow (L)
u^* = shear velocity (L/T).

In turn, the following equation estimates shear velocity:

$$u^* = (gds)^{1/2}$$

where: g = acceleration due to gravity (L/T^2)
s = slope of channel (L/L)
d = water depth (L).

The model DYNTOX (LimnoTech, 1985) is specially designed for analysis of toxics in rivers and can handle all three dynamic modeling techniques. U.S. EPA (1991a) and the WLA series by Delos et al. (1984) address the transport of toxics and heavy metals in rivers.

Oxygen Demand/Dissolved Oxygen. The time scales and distances affecting DO processes are greater than for bacteria and toxics. Lateral mixing therefore results in approximately uniform conditions over the river cross section and one-dimensional models are usually appropriate for simulation. The WLA guidance (U.S. EPA, 1995g) discusses the effects of steady and dynamic DO loads, and provides guidelines for modeling impacts of steady-state sources. Simple spreadsheet models such as STREAMDO IV (Zander and Love, 1990) have recently become available for DO analysis.

In general, screening analyses using classical steady-state equations can examine DO impacts to rivers as a result of episodic loads. This approach assumes plug flow, which in turn allows an assumption of constant loading averaged over the volume of the plug (Freedman and Marr, 1990). This approach does not consider longitudinal diffusion from the plug, making it a conservative approach. The plug flow analysis should correlate with the duration of the CSO. For example, a plug flow simulation of a 2-hour CSO event would result in a downstream DO sag that would also last for 2 hours. Given the plug flow assumption, the classic Streeter-Phelps equation can estimate the DO concentration downstream:

$$D = D_o e^{-K_a t} + \frac{W}{Q}\left(\frac{K_d}{K_a - K_r}\right)\left[e^{-K_r t} - e^{-K_a t}\right]$$

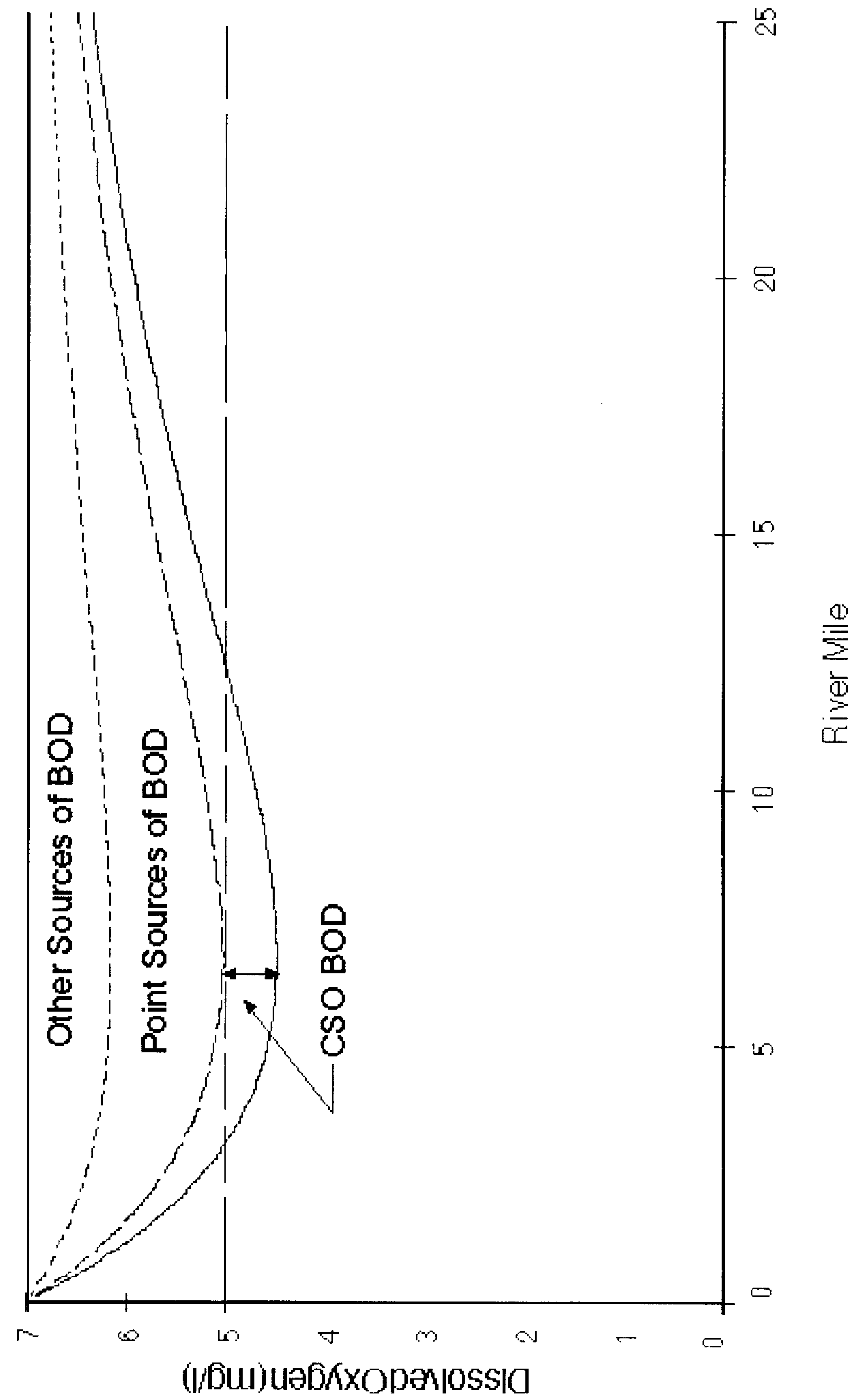
where: D = DO deficit downstream (M/V)
D_0 = initial DO deficit (M/V)
K_a = atmospheric re-aeration rate (1/T)
t = time of passage from source to downstream location (T)
W = total pollutant loading rate (M/T)
Q = total river flow (V/T)
K_d = biochemical oxygen demand (BOD) deoxygenation rate (1/T)
K_r = BOD loss rate (1/T).

This method can address the joint effects of multiple steady sources through the technique of superposition (Exhibit 8-1). Superposition is used when linear differential equations, such as the Streeter-Phelps equation, govern pollutant concentrations along a receiving stream. For such linear systems, the concentration of a pollutant in a river due to multiple steady-state sources is the linear summation of the responses due to the individual sources. Superposition techniques are also used to estimate pollutant concentrations due to multiple steady-state sources of toxic pollutants. However, it cannot address multiple sources that change over time, nor can it address the effects of river morphology. When such issues are important, more sophisticated modeling techniques are necessary.

More sophisticated modeling techniques are also necessary to assess the effects of sediment oxygen demand (SOD) and plant respiration (which remove oxygen from the receiving water), and photosynthesis by aquatic plants (which adds oxygen to the water). The Streeter-Phelps equation makes the simplifying assumption that there are only point sources of CBOD, so SOD, photosynthesis, and respiration are assumed to be zero. If photosynthesis, respiration, and SOD are significant, more complex analysis is needed to evaluate these factors. These distributed sources and sinks of DO and BOD are addressed by Thomann and Mueller (1987) and by several computer models, including QUAL2EU and WASPS.

Nutrients/Eutrophication. Nutrient discharges affect river eutrophication over time scales of several days to several weeks. Nutrient/eutrophication analysis considers the relationship between

Exhibit 8-1. Dissolved Oxygen Superposition Analysis



nutrients and algal growth. Analysis of nutrient impacts in rivers is complex because nutrients and planktonic algae,[3] which are free-floating one-celled algae, usually move through the system rapidly.

The current WLA guidance (U.S. EPA, 1995g) considers only planktonic algae (rather than all aquatic plants) and discusses nutrient loadings and eutrophication in rivers primarily as a component in computing DO. The guidance applies to narrative criteria that limit nuisance plant growth in large, slowly flowing rivers.

LAKES

Bacteria and Toxics. Mixing zone analysis can often be used to assess attainment of WQS for bacteria and toxics in lakes. For a small lake in which the effluent mixes rapidly, the concentration response is given by the following equation (Freedman and Marr, 1990):

$$C = \frac{M}{V} e^{(-K - \frac{Q}{V})t}$$

where: C = concentration (M/L^3)
M = mass loading (M)
Q = flow (L^3/T)
K = net decay rate (bacteria die-off, settling and resuspension, volatilization, photolysis, and other chemical reactions) (1/T)
V = lake volume (L^3)
t = time (T).

For an incompletely-mixed lake, however, a complex simulation model is generally necessary to estimate transient impacts from slug loads. The EPA WLA guidance series contains a manual on chemical models for lakes and impoundments (Hydroqual, Inc., 1986). This guidance, which also applies to bacteria, describes simple and complex models and presents criteria for selecting models and model parameters.

[3] Aquatic plants can be divided into those that move freely with the water (planktonic aquatic plants) and those that are attached or rooted in place.

Oxygen Demand/Dissolved Oxygen. Simple analytical approximations can model oxygen demand and DO in cases where DO mixing occurs quickly relative to depletion by COD/BOD. Where lateral mixing occurs rapidly but vertical temperature stratification exists, DO concentration can be addressed for a two-layer stratified lake under the following simplifying assumptions (from Thomann and Mueller, 1987):

- The horizontal area is constant with depth
- Inflow occurs only to the surface layer
- Photosynthesis occurs only in the surface layer
- Respiration occurs at the same rate throughout the lake
- The lake is at steady-state.

With these severe restrictions, the solution is given by:

$$c_1 = \left(\frac{q}{K_L + q}\right)c_o + \left(\frac{K_L}{K_L + q}\right)c_s + \frac{pH_1 - RH - S_B}{K_L + q} - \frac{K_{d1}H_1L_1 - K_{d2}H_2L_2}{K_L + q}$$

and

$$c_2 = c_1 - \left(\frac{S_B + RH_2 - K_{d2}H_2L_2}{E/H_i}\right)$$

where the subscripts 1 and 2 refer to the epilimnion (top layer) and hypolimnion (lower layer), respectively, and variables without subscripts refer to the whole lake, and where:

q	=	Outflow rate (L/T)
K_L	=	DO transfer rate at lake surface (L/T)
c	=	DO concentration (M/L^3)
c_0, c_s	=	Initial and saturation dissolved oxygen concentrations (M/L^3)
p	=	Gross photosynthetic production of DO (m/L^3-T)
H	=	Depth (L)
H_i	=	$H/2$ when $H_1 = H_2$ and H_1 when $H_2 >> H_1$ (L)
R	=	Phytoplankton DO respiration (M/L^3-T)

S_B = Sediment oxygen demand (M/L^2-T)
K_d = Deoxygenation coefficient (1/T)
L = Steady-state CBOD concentration in water column (M/L^3), = $W/(Q+K_rV)$, where W is the mass loading rate, Q is the rate of flow through the lake, V is the volume, and K_r is the net loss rate.
E = Dispersion coefficient (L^2/T).

Because this analysis assumes steady-state loading and because measuring some of the parameters proves difficult, the method may only have limited application to CSOs. A modeler able to define all of the above parameters may choose to apply a more spatially resolved model.

In many cases, complex simulation models are necessary to analyze DO in lakes. These are either specialized lake models or flexible models, such as EUTROWASP, that are designed to address issues specific to lakes. Some experienced modelers have been successful in modeling thermally stratified lakes with one or two dimensional river models (e.g., QUAL2EU) that assume the river bottom is the thermocline.[4]

Nutrient/Eutrophication Impacts. For lakes, simple analytic equations often can analyze end-of-pipe impacts and whole-lake impacts, but evaluating mixing phenomena frequently requires a complex computer model (Freedman and Marr, 1990). Simple analytical methods can be applied to lake nutrient/eutrophication impacts in situations where the CSOs mix across the lake area within the time scale required to obtain a significant response in the algal population. In most lakes, phosphorus is considered to be the limiting nutrient for nuisance algal impacts and eutrophication. Mancini et al. (1983) and Thomann and Mueller (1987) have developed a procedure for calculating the allowable surface loading rate. The following steps are drawn from this procedure:

Step 1. Estimate the lake volume, surface area, and mean depth.

Step 2. Estimate the mean annual inflow and outflow rates. Where urban areas draining to the lake constitute a significant fraction of the total drainage area, flow

[4] Such techniques should not be used by inexperienced modelers as they can lead to inaccuracies if they are not used with caution.

estimates from urban runoff and CSOs should be included in the hydrologic balance around the lake. For lakes with large surface areas, the estimate should include surface precipitation and evaporation.

Step 3. Determine the average annual total phosphorus loading due to all sources, including all tributary inflows, municipal and industrial sources, distributed urban and rural runoff, and atmospheric inputs. ***Technical Guidance Manual for Performing Waste Load Allocation*** (Mancini et al., 1983) discusses techniques for estimating these loadings.

Step 4. For total phosphorus, assign a net sedimentation loss rate that is consistent with a local data base.

Step 5. Select trophic state objectives of either total phosphorus or chlorophyll-a consistent with local experience. Calculate the value of the allowable phosphorus area1 loading, W^{1}, from:

$$W' = a\bar{z}\left(\frac{Q}{V} + v_s\right)$$

where: W' is the allowable area1 surface loading rate (M/L^2-T)
a is the trophic state objective concentration of total phosphorus or chlorophyll-a (M/L^3),
Q is outflow (L^3/T),
V is lake volume (L^3),
$\bar{z}$ is mean depth (L), and
v_s is the net sedimentation velocity (L/T).

Step 6. Compare the total area1 loading determined in Step 3 to the value of W^{1} obtained in Step 5.

Additional approaches are discussed in Reckhow and Chapra (1983b).

ESTUARIES

Unlike most rivers, estuaries are tidal (i.e., water moves upstream during portions of the tidal cycle and downstream during other parts of the cycle). When averaged on the basis of tidal cycles, pollutant transport in narrow, vertically mixed estuaries with dominant longitudinal flow is similar to that in rivers. However, due to tidal reversals of flow, a narrow estuary may have a much larger effective dispersion coefficient since shifting tides may cause greater lateral dispersion. In such a system, the modeler can apply approximate or screening models used for rivers, provided that an appropriate tidal dispersion coefficient has been calculated. In wider estuaries, tides and winds often result in complex flow patterns and river-based models would be inappropriate. WLA guidance for estuaries is provided in several EPA manuals (Ambrose et al., 1990; Martin et al., 1990; Jirka, 1992; Freedman et al., 1992).

In addition to their tidal component, many estuaries are characterized by salinity-based stratification. Stratified estuaries have the horizontal mixing due to advection and dispersion that is associated with rivers and the vertical stratification characteristic of lakes.

In complex estuaries, accurate analysis of far-field CSO impacts-such as nutrients/ eutrophication, DO, and impacts on particular sensitive areas-typically requires complex simulation models. Simpler analyses are sometimes possible by treating the averaged effects of tidal and wind-induced circulation and mixing as temporally constant parameters. This approach may require extensive site-specific calibration.

Near-field mixing zone analysis in estuaries also presents special problems, because of the role of buoyancy differences in mixing. Jirka (1992) discusses mixing-zone modeling for estuaries.

8.3.2 Computer Models Supported by EPA or Other Government Agencies

This section describes some computer models relevant to receiving water modeling. Most of these models are supported by EPA's Center for Exposure Assessment Modeling (CEAM). CEAM maintains a distribution center for water quality models and related data bases.[5] CEAM-supported models relevant to modeling impacts on receiving water include QUAL2EU, WASPS, HSPF, EXAMSII, CORMIX, MINTEQ, and SMPTOX3. The applicability and key characteristics of the CEAM-supported models are summarized in Exhibit 8-2.

[5] See Section 7.3 for information on obtaining models from CEAM.

Exhibit 8-2. EPA CEAM-Supported Receiving Water Models

Applicability to Hydraulic Regimes and Pollutant Type										
	Rivers & Streams			Lakes & Impoundments			Estuaries			Near Field Mixing
Model	**Nutrients**	**Oxygen**	**Other**	**Nutrients**	**Oxygen**	**Other**	**Nutrients**	**Oxygen**	**Other**	
QUAL2EU	✓	✓	✓							
WASP5	✓	✓	✓	✓	✓	✓	✓	✓	✓	
HSPF	✓	✓	✓	✓	✓	✓				
EXAMSII			✓			✓			✓	
CORMIX	Near-field mixing model for all water body types									✓
MINTEQ	Equilibrium metal speciation model									
SMPTOX3			✓							

Key Characteristics and References				
Model	**Pollutant Loading Type**	**Transport Dimensionality**	**Current Version**	**Key References**
QUAL2EU	Steady	1-D	3.22	Brown & Barnwell, 1987
WASP5	Dynamic	Quasi-2/3-D (link-node)	5.10	Ambrose, et al., 1988
HSPF	Dynamic (integrated)	1-D	10.11	Johanson, et al., 1984
EXAMSII	Dynamic	User input (quasi 3-D)	2.96	Burns, et al., 1982
CORMIX	Steady (near field) [1]	Quasi-3-D (zonal)	2.10	Doneker & Jirka, 1990
MINTEQ	Steady	None	3.11	Brown & Allison, 1987
SMPTOX3	Steady	1-D	2.01	LimnoTech, 1992

[1] CORMIX was originally developed assuming steady ambient conditions; Version 3 allows for application to some unsteady environments (e.g., tidal reversal conditions) where transient recirculation and pollutant build-up can occur (CEAM, 1998).

QUAL2EU is a one-dimensional model for rivers. It assumes steady-state flow and loading but allows simulation of diurnal variations in temperature or algal photosynthesis and respiration. QUAL2EU simulates temperature, bacteria, BOD, DO, ammonia, nitrate, nitrite, organic nitrogen, phosphate, organic phosphorus, algae, and additional conservative substances.[6] Because it assumes steady flow and pollutant loading, its applicability to CSOs is limited. QUAL2EU can, however, use steady loading rates to generate worst-case projections for CSOs to rivers. The model has pre- and post-processors for performing uncertainty and sensitivity analyses.

Additionally, in certain cases, experienced users may be able to use the model to simulate non-steady pollutant loadings under steady flow conditions by establishing certain initial conditions or by dynamically varying climatic conditions. If used in this way, QUAL2EU should be considered a screening tool since the model was not designed to simulate dynamic quality conditions.

WASP5 is a quasi-two-dimensional or quasi-three-dimensional water quality model for rivers, estuaries, and many lakes. It has a link-node formulation, which simulates storage at the nodes and transport along the links. The links represent a one-dimensional solution of the advection dispersion equation, although quasi-two-dimensional or quasi-three-dimensional simulations are possible if nodes are connected to multiple links. The model also simulates limited sediment processes. It includes the time-varying processes of advection, dispersion, point and nonpoint mass loading, and boundary exchanges. WASP5 can be used in two modes: EUTRO5 for nutrient and eutrophication analysis and TOXI5 for analysis of toxic pollutants and metals.

WASP5 is essentially a pollutant fate and transport model. Transport can be driven by another hydrodynamic model such as DYNHYD5. DYNHYD5 is a one-dimensional/quasi-two-dimensional model that simulates transient hydrodynamics (including tidal estuaries).

[6] A conservative substance is one that does not undergo any chemical or biological transformation or degradation in a given ecosystem. (U.S. EPA, 1995g)

HSPF is a one-dimensional, comprehensive hydrologic and water quality simulation package which can simulate both receiving waters and runoff to CSSs for conventional and toxic organic pollutants. HSPF simulates the transport and fate of pollutants in rivers and reservoirs. It simulates three sediment types: sand, silt, and clay.

EXAMSII can rapidly evaluate the fate, transport, and exposure concentrations of steady discharges of synthetic organic chemicals to aquatic systems. A recent upgrade of the model considers seasonal variations in transport and time-varying chemical loadings, making it quasi-dynamic. The user must specify transport fields to the model.

CORMIX[7] is an expert system for mixing zone analysis. It can simulate submerged or surface, buoyant or non-buoyant discharges into stratified or unstratified receiving waters, with emphasis on the geometry and dilution characteristics of the initial mixing zone. The model uses a zone approach, in which a flow classification scheme determines which near-field mixing processes to calculate. The CORMIX model cannot be calibrated in the classic sense since rates are fixed based on the built-in logic of the expert system.

MINTEQ determines geochemical equilibrium for priority pollutant metals. Not a transport model, MINTEQ provides a means for modeling metal partitioning in discharges. It provides only steady-state predictions. The model usually must be run in connection with another fate and transport model, such as those described above. A number of assumptions (e.g., equilibrium conditions at the point of mixing between a CSO and the receiving water) must be made to link MINTEQ predictions to another fate and transport model, so it should be used cautiously in evaluating wet weather impacts.

SMPTOX3 is a one-dimensional steady-state model for simulating the transport of contaminants in the water column and bed sediments in streams and non-tidal rivers. SMPTOX3 is an interactive computer program that uses an EPA technique for calculating concentrations of

[7] In some applications CORMIX has proven inaccurate for single port discharges.

toxic substances in the water column and stream bed as a result of point source discharges to streams and rivers. The model predicts pollutant concentrations in dissolved and particulate phases for the water column and bed sediments, as well as total suspended solids. SMPTOX3 can be run at three different levels of complexity: as described above (highest complexity), to calculate toxic water column concentrations but no interactions with bed sediments (medium complexity), or as a total pollutant toxics model (low complexity) (LimnoTech, 1992).

The following additional models are supported by EPA or other government agencies:[8]

DYNTOX is a one-dimensional, probabilistic toxicity dilution model for transport in rivers. It provides continuous, Monte Carlo, or lognormal probability simulations that can be used to analyze the frequency and duration of ambient toxic concentrations resulting from a waste discharge. The model considers dilution and net first-order loss, but not sorption and benthic exchange. DYNTOX Version 2.1 and the draft manual are available from the Office of Science and Technology in EPA's Office of Water (202-260-7012).

CE-QUAL-W2 is a reservoir and narrow estuary hydrodynamics and water quality model developed by the Waterways Experiment Station of the U.S. Army Corps of Engineers. The model provides dynamic two-dimensional (longitudinal and vertical) simulations. It accounts for density effects on flow as a function of the water temperature, salinity and suspended solids concentration. CE-QUAL-W2 can simulate up to 21 water quality parameters in addition to temperature, including one passive tracer (e.g., dye), total dissolved solids, coliform bacteria, inorganic suspended solids, algal/nutrient/DO dynamics (11 parameters), alkalinity, pH and carbonate species (4 parameters).

[8] McKeon and Segna (1987), Ambrose et al. (1988a) and Hinson and Basta (1982) have reviewed some of these models.

8.4 USING A RECEIVING WATER MODEL

As was the case for CSS models (see Section 7.4), receiving water modeling involves developing the model, calibrating and validating the model, performing the simulation, and interpreting the results.

8.4.1 Developing the Model

The input data needs for a specific receiving water model depend upon the hydraulic regime and model used. The permittee should refer to the model's documentation, the relevant sections of the WLA guidance, or to texts such as *Principles of Surface Water Quality Modeling and Control* (Thomann and Mueller, 1987). Tables B-2 through B-5 in Appendix B contain general tables of data inputs.

8.4.2 Calibrating and Validating the Model

Like CSS models, receiving water models need to be calibrated and validated. The model should be run to simulate events for which receiving water hydraulic and quality monitoring were actually conducted, and the model results should be compared to the measurements. Generally, receiving water models are calibrated and validated first for receiving water hydraulics and then for water quality. Achieving a high degree of accuracy in calibration can be difficult because:

- Pollutant loading inputs typically are estimates rather than precisely known values.
- Three-dimensional receiving water models are still not commonly used for CSO projects, so receiving water models involve spatial averaging (over the depth, width or cross-section). Thus, model results are not directly comparable with measurements, unless the measurements also have sufficient spacial resolution to allow comparable averaging.
- Loadings from non-CSO sources, such as storm water, upstream boundaries, point sources, and atmospheric deposition, often are not accurately known.
- Receiving water hydrodynamics are affected by numerous factors which are difficult to account for. Those include fluctuating winds, large-scale eddies, and density effects.

Although these factors make model calibration challenging, they also underscore the need for calibration to ensure that the model reasonably reflects receiving water data.

8.4.3 Performing the Modeling Analysis

Receiving water modeling can involve single events or long-term simulations. Single event simulations are usually favored when using complex models, which require more input data and take significantly longer to run (although advances in computer technology keep pushing the limits of what can practically be achieved.) Long-term simulations can predict water quality impacts on an annual basis.

Although a general goal is to predict the number of water quality criteria exceedances, models can evaluate exceedances using different measures, such as hours of exceedance at beaches or other critical points, acre-hours of exceedance, and mile-hours of exceedance along a shore. These provide a more refined measure of the water quality impacts of CSOs and of the expected effectiveness of different control measures.

CSO loadings commonly are simulated separately from other loadings in order to assess the relative impacts of CSOs. This is appropriate because the equations that best approximate receiving water quality are usually linear and so effects are additive (one exception, however, is the non-linear algal growth response to nutrient loadings).

8.4.4 Using Modeling Results

By calculating averages over space and time, simulation models predict CSO volumes, pollutant concentrations, and other variables of interest. The extent of this averaging depends on the model structure, how the model is applied, and the resolution of the input data. The model's space and time resolution should match that of the necessary analysis. For instance, the applicable WQS may be expressed as a 1-hour average concentration not to exceed a given concentration more than once every three years on average. Spatial averaging may be represented by a concentration averaged over a receiving water mixing zone, or implicitly by the specification of monitoring locations to establish compliance with instream criteria. In any case, the permittee should note whether the model predictions use the same averaging scales required in the permit or relevant WQS.

When used for continuous rather than event simulation, as suggested by the CSO Control Policy, simulation models can predict the frequency of exceedances of water quality criteria. Probabilistic models, such as the Monte Carlo simulation, also can make such predictions. In probabilistic models, the simulation is made over the probability distribution of precipitation and other forcing functions such as temperature, point sources, and flow. In either case, modelers can analyze the output for the frequency of water quality criteria exceedances.

The key result of receiving water modeling is the prediction of future conditions due to implementation of CSO control alternatives. In most cases, CSO control decisions will have to be supported by model predictions of the pollutant load reductions necessary to achieve WQS. In the receiving waters, critical or design water quality conditions might be periods of low flows and high temperature that are established based on a review of available data. Flow, temperature, and other variables for these periods then form the basis for analysis of future conditions.

It is useful to assess the sensitivity of model results to variations in parameters, rate constants, and coefficients. A sensitivity analysis can determine which parameters, rate constants, and coefficients merit particular attention in evaluating CSO control alternatives. The modeling approach should accurately represent features that are fully understood, and sensitivity analysis should be used to evaluate the significance of factors that are not as clearly defined. (See Section 7.4.4 for additional discussion of sensitivity analysis.)

CHAPTER 9

ASSESSING RECEIVING WATER IMPACTS AND ATTAINMENT OF WATER QUALITY STANDARDS

This chapter focuses on the link between CSOs and the attainment of water quality standards (WQS). As discussed in previous chapters, permittees can consider a variety of methods to analyze the performance of the combined sewer system (CSS) and the response of a water body to pollutant loads. Permittees can use these methods to estimate the water quality impacts of a proposed CSO control program and evaluate whether it is adequate to meet CWA requirements.

Under the CSO Control Policy, permittees need to develop long-term control plans (LTCPs) that provide for WQS attainment using either the presumption approach or the demonstration approach. This chapter focuses primarily on issues related to the demonstration approach since this approach requires the permittee to demonstrate that the selected CSO controls will provide for the attainment of WQS. As mentioned in Chapter 8, the presumption approach does not explicitly call for analysis of receiving water impacts and thus generally involves less complex modeling.

Modeling time-varying wet weather sources such as CSOs is more complex than modeling more traditional point sources. Typically, point-source modeling assumes constant pollutant loading to a receiving water body under critical, steady-state conditions-such as the minimum seven-consecutive-day average stream flow occurring once every ten years (i.e., 7Q10). Wet weather loads occur in pulses, however, and often have their peak impacts under conditions other than low-flow situations. This makes modeling the in-stream impact of CSOs more complicated than modeling the impacts of steady-state point source discharges such as POTWs. A receiving water model must therefore accommodate the short-term variability of pollutant concentrations and flow volume in the discharge as well as the dynamic conditions in the receiving water body. Notwithstanding these limitations, however, properly-applied modeling techniques can be useful in analyzing the impact of CSOs on receiving waters.

CSO pollutant loads can be incorporated into receiving water models using either a steady-state or a dynamic approach, as discussed in Chapter 8. A steady-state model can provide an approximate solution using, for example, average loads for a design storm. A dynamic approach incorporates time-varying loads and simulates the time-varying response of the water body. The steady-state approximation uses some average conditions that do not account for the time-varying nature of flows and loads. Thus a steady-state model may provide less exact results, but typically requires less cost and effort. A dynamic model requires more resources but may result in a more cost-effective CSO control plan, since it does not use some of these simplifying assumptions.

Generally, the modeler should use the simplest approach that is appropriate for local conditions. A steady-state model may be appropriate in a receiving water that is relatively insensitive to short-term variations in load rate. For instance, the response time of lakes and coastal embayments to some pollutant loadings may be measured in weeks to years, and the response time of large rivers to oxygen demand may be measured in days (Donigian and Huber, 1991). Steady-state models are also useful for estimating the dilution of pollutants, such as acute toxins or bacteria, close to the point of release.

9.1 IDENTIFYING RELEVANT WATER QUALITY STANDARDS

The demonstration approach requires the permittee to show that its selected CSO controls will provide for attainment of WQS. The CSO Control Policy states that:

> *The permittee should demonstrate...*
>
> *i. the planned control program is adequate to meet WQS and protect designated uses, unless WQS or uses cannot be met as a result of natural background conditions or pollution sources other than CSOs;*
>
> *ii. the CSO discharges remaining after implementation of the planned control program will not preclude the attainment of WQS or the receiving waters' designated uses or contribute to their impairment. Where WQS and designated uses are not met in part because of natural background conditions or pollution sources other than CSOs, a total maximum daily load, including a wasteload allocation and a load allocation, or other means should be used to apportion pollutant loads...* (Section II.C.4.b)

The first step in analyzing CSO impacts on receiving water is to identify the pollutants or stressors of concern and the corresponding WQS. CSOs are distinguished from storm water loadings by the increased levels of such pollutants as bacteria, oxygen-demanding wastes, and certain nutrients. In some cases, toxic pollutants entering the CSS from industrial sources also may be of concern.

State WQS include designated uses and both numerical and narrative water quality criteria. Since CSO controls must ultimately provide for attainment of WQS, the analysis of CSO control alternatives should be tailored to the applicable WQS. For example, if the water quality criterion of concern is expressed as a daily average concentration, the analysis should address daily averages. Many water bodies have narrative criteria such as a requirement to limit nutrient loads to an amount that does not produce a "nuisance" growth of algae, or a requirement to prevent solids and floatables build-up. In such cases, the permittee could consider developing a site-specific, interim numeric performance standard that would result in attainment of the narrative criterion.

As noted in Chapter 2, a key principle of the CSO Control Policy is the review and revision, as appropriate, of WQS and their implementation procedures. In identifying applicable WQS, the permittee and the permitting and WQS authorities should consider whether revisions to WQS are appropriate for wet weather conditions in the receiving water.

EPA's water quality criteria assist States in developing numerical standards and interpreting narrative standards (U.S. EPA, 1991a). EPA recommends that water quality criteria for protection of aquatic life have a magnitude-duration-frequency format, which requires that the concentration of a given constituent not exceed a critical value more than once in a given return period:

- ***Magnitude-*** The concentration of a pollutant, or pollutant parameter such as toxicity, that is allowable.
- ***Duration-*** The averaging period, which is the period of time over which the in-stream concentration is averaged for comparison with criteria concentrations. This specification limits the duration of concentrations above the criteria.

- ***Frequency-*** How often criteria can be exceeded.

A magnitude-duration-frequency criteria statement directly addresses protection of the water body by expressing the acceptable likelihood of excursions above the WQS. Although this approach appears useful, it requires estimation of long-term average rates of excursion above WQS.

Many States rely instead on the concept of design flows, such as 7Q10. Evaluating compliance at a design low flow of specified recurrence is a simple way to approximate the average duration and frequency of excursions above the WQS. A single critical low flow, however, is not necessarily the best choice for wet-weather flows, which may not occur simultaneously with drought conditions. Consequently, a design flow-based control strategy may be overly conservative, and suitable mainly for situations where monitoring data are very limited or areas are highly sensitive.

Some water quality criteria are expressed in formats that vary from the magnitude-duration-frequency format. In some cases, such as State WQS for indicator bacteria, water quality criteria are expressed as an instantaneous maximum and a long-term average component. The long-term average component of water quality criteria for fecal coliforms typically specifies a 30-day geometric mean or median, and a certain small percentage of tests performed within a 30-day period that may exceed a particular upper value. For dissolved oxygen (DO) and pH, State criteria may be expressed as fixed minimum concentrations, rather than as magnitude-duration-frequency.

The statistical form of the relevant WQS is important in determining an appropriate model framework. Does the permittee need to calculate a long-term average, a worst case maximum, or an actual time sequence of the number of water quality excursions? An approach that gives a reasonable estimate of the average may not prove useful for estimating an upper bound.

9.2 OPTIONS FOR DEMONSTRATING COMPLIANCE

Receiving water impacts can be analyzed at varying levels of complexity, but all approaches attempt to answer the same question: *Using a prediction of the frequency and volume of CSO events and the pollutant loads delivered by these events, can WQS in the receiving water body be attained with a reasonable level of assurance?*

Any of the following types of analyses, arranged in order of increasing complexity, can be used to answer this question:

- ***Design Flow Analysis-*** This approach analyzes the impacts of CSOs under the assumption that they occur at a design condition (e.g., 7Q10 low flow prior to addition of the CSO flow). The CSO is added as a steady-state load. If WQS can be attained under such a design condition, with the CSO treated as a steady source, WQS are likely to be attained for the actual wet weather conditions. This approach is conservative in two respects: (1) it does not account for the short-term pulsed nature of CSOs, and (2) it does not account for increased receiving water flow during wet weather.

- ***Design Flow Frequency Analysis-*** Where the WQS is expressed in terms of frequency and duration, the frequency of occurrence of CSOs can be included in the analysis. The design flow approach can then be refined by determining critical design conditions that can reasonably be expected to take place concurrently with CSOs. For instance, if CSO events occur primarily in one season, the analysis can include critical flows and other conditions appropriate to that season, rather than the 7Q10.

- ***Statistical Analysis-*** Whereas the previous two approaches rely on conservative design conditions, a statistical analysis can be used to consider the range of flows that may occur together with CSO events. This analysis more accurately reflects the frequency of WQS excursions.

- ***Watershed Simulation-*** A statistical analysis does not consider the dynamic relationship between CSOs and receiving water flows. For example, both the CSO and receiving water flows increase during wet weather. Demonstrating the availability of this additional capacity, however, requires a model that includes the responses of both the sewershed and its receiving water to the rainfall events. Dynamic watershed simulations may be carried out for single storm events or continuously for multiple storm events.

The permittee should consider the tradeoffs between simpler and more complex types of receiving water analysis. A more complex approach, although more costly, can generally provide more precise analysis using less conservative assumptions. This may result in a more tailored, cost-effective CSO control strategy.

Additional discussion on data assessment for determining WQS attainment is in *Guidelines for the Preparation of the 1996 State Water Quality Assessments (305(b) Reports)* (U.S. EPA, 1995f).

9.3 EXAMPLES OF RECEIVING WATER ANALYSIS

This section presents three examples to illustrate key points for analyzing CSO impacts on receiving waters. The examples focus on (1) establishing the link between model results and demonstrating the attainment of WQS, and (2) the uses of receiving water models at different levels of complexity, from design flow analysis to dynamic continuous simulation.

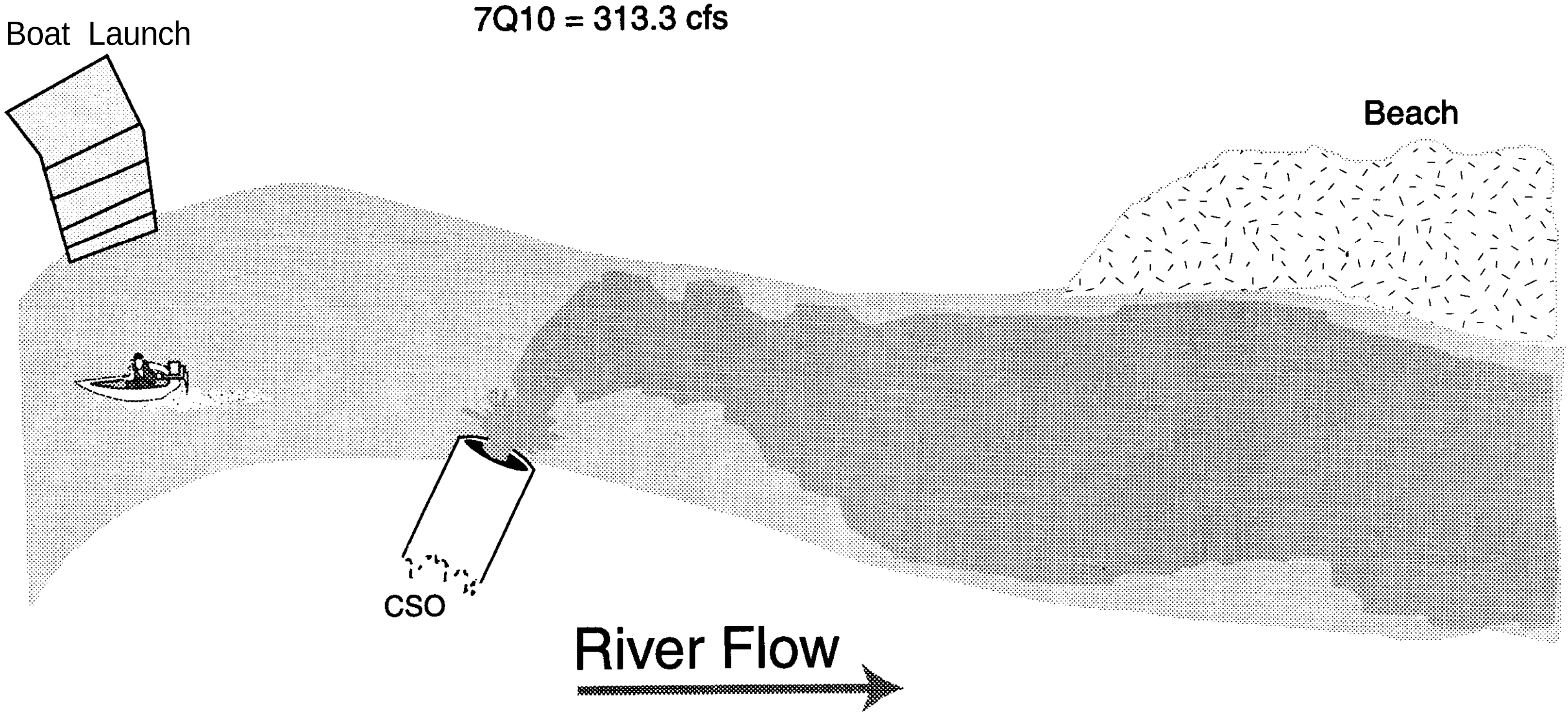
The first example shows how design flow analysis or more sophisticated methods can be used to analyze bacteria loads to a river from a single CSO event. The second example, which is more complex, involves bacterial loads to an estuary. The third example illustrates how biochemical oxygen demand (BOD) loads from a CSS contribute to DO depletion.

9.3.1 Example 1: Bacterial Loads to a River

This example involves a CSS in a small northeastern city that overflows relatively frequently and contributes to WQS excursions. CSOs are the only pollutant source, and only a single water quality criterion--for fecal coliforn-applies. The use classification for this receiving water body is primary and secondary contact recreation. The city has planned several engineering improvements to its CSS and wishes to assess the water quality impacts of those improvements.

Exhibit 9-1 is a map of key features in this example.

Exhibit 9-1. Map For Example 1



In this example, dilution calculations may suffice to predict whether the water quality criterion is likely to be attained during a given CSO event. This is because:

(1) The State allows mixing zones, so the water quality criterion must be met at the edge of the mixing zone. If the criterion is met there, it will also be met at points farther away.

(2) Die-off will reduce the numbers of bacteria as distance from the discharge increases.

(3) Since the river flows constantly in one direction, bacterial concentrations do not accumulate or combine loads from several days of release.

To illustrate the various levels of receiving water analysis, this example assumes that the magnitude and timing of CSOs can be predicted precisely and that the long-term average characteristics of the CSS will remain constant. In the absence of additional CSO controls, the predictions for the next 31 years include the following (Exhibit 9-2):

(1) The system should experience a total of 238 overflow events, an average of 7.7 per year.[1]

(2) The largest discharge is approximately 1.1 million cubic feet, but most of the CSOs are less than 200,000 cubic feet.

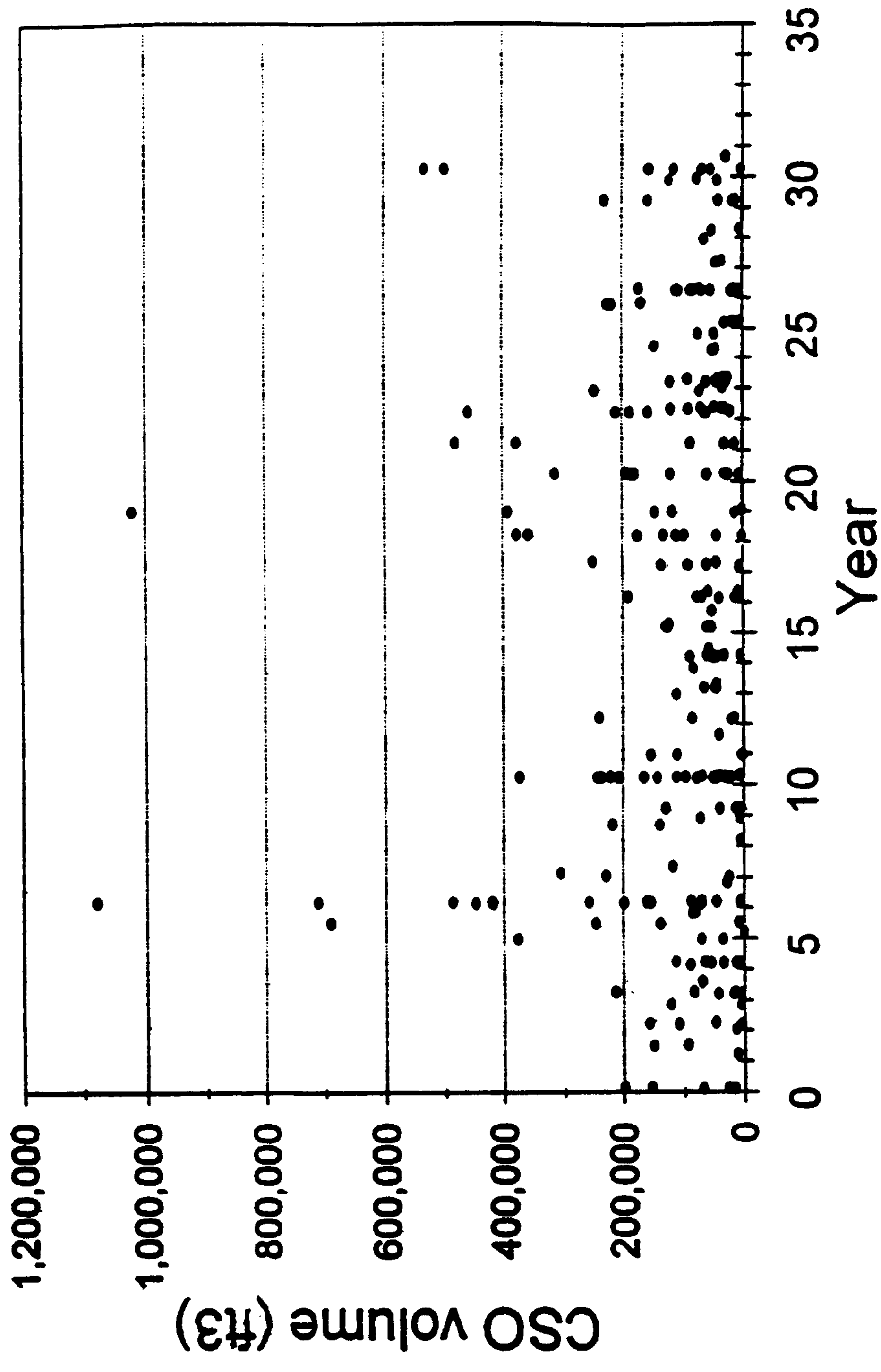
(3) The maximum number of overflow events in any one month is 18.

(4) During that month, the maximum receiving water concentration resulting from CSOs exceeds 6,000 MPN/100 ml. Even in this "worst-case" month, however, the geometric mean is 400 MPN/100 ml, based on 30 daily samples and assuming a background concentration of 100.

At least one CSO event occurs in each calendar month, although 69 percent of the events occur in March and April when snowmelt increases flow in the CSS. Because river flow is lower in summer and fall, the rarer summer and fall CSOs may cause greater impact in the receiving water.

[1] An overflow event is the discharge from one or more CSO outfalls as the result of a single wet weather event. In this example, the number and volume of CSOs pertains to the discharges from the single outfall.

Exhibit 9-2. CSO Events for Example 1



Water Quality Standards

The applicable water quality criterion for fecal coliforms specifies that:

(1) The geometric mean for any 30-day period not exceed 400 MPN (“most probable number”) per 100 ml, and

(2) Not more than 10 percent of samples taken during any 30-day period exceed 1,000 MPN per 100 ml.[2]

The water quality criterion does not specify an instantaneous maximum count for this use classification.

It is comparatively simple to assess how the first component-the geometric mean of 400 MPN/100 ml-applies.[3] In the worst-case month, which had 18 overflow events, the geometric mean is still only 400 MPN/100 ml based on 30 daily samples. It is therefore extremely unlikely that the geometric mean concentration WQS of 400 MPN/100 ml will be violated in any other month.

In general, the second component of the water quality criterion-a percentile (or maximum) standard-will prove more restrictive for CSOs. A CSS that overflows less than 10 percent of the time (fewer than 3 days per month) could be expected to meet a not-more-than-10-percent requirement, *on average*, but probably only if loads from other sources were well below 1000 MPN/100 ml and the CSS discharged to a flowing river system, where bacteria do not accumulate from day to day. It is possible that an actual overflow event might not result in an excursion above the 1000 MPN/100 ml criterion *if* the flow in the receiving water were sufficiently large. The permittee, however, must demonstrate that the likelihood of a 30-day period when CSOs result in non-attainment of the WQS more than 10 percent of the time is *extremely low*. This means that the analysis must consider both the likelihood of occurrence of overflow events and the dilution

[2] Most Probable Number (MPN) of organisms present is an estimate of the average density of fecal coliforms in a sample, based on certain probability formulas.

[3] The geometric mean, which is defined as the antilog of the average of the logs of the data, typically approximates the median or midpoint of the data.

capacity of the receiving water at the time of an overflow. The following sections demonstrate various ways to make this determination.

Design Flow Analysis

Design flow analysis is the simplest but not necessarily the most appropriate approach. It uses conservatively low receiving water flow to represent the minimum reasonable dilution capacity. If the effects of all CSO events would not prevent the attainment of WQS under these stringent conditions, the permittee has clearly demonstrated that the applicable WQS should be attained. In cases where nonattainment is indicated, however, the necessary reductions to reach attainment may be unreasonably high since CSOs are unlikely to occur at the same time as design low flows.

The CSO outfall in this example is at a bend in the river where mixing is rapid. Therefore, the loads are considered fully mixed through the cross-section of flow. The concentration in the receiving water is determined by a simple mass balance equation,

$$C_{RW} = \frac{C_{CSO} Q_{CSO} + C_U Q_U}{Q_{CSO} + Q_U}$$

where C represents concentration and Q flow (in any consistent units). The subscripts RW, CSO, and U refer to “receiving water,” “combined sewer overflow,” and “upstream,” respectively.

For the design flow analysis, upstream volume Q_u is set to a low flow of specified recurrence and receiving water concentration C_{RW} is set equal to the water quality criterion. In this example, upstream volume Q_U is set at the 7Q10 flow. The 7Q10 flow is commonly used for steady-state wasteload analyses; although it has a lo-year recurrence and is much more stringent than the not-more-than-10-percent requirement of the standard, this conservatism ensures that excursions of the standard will indeed occur only rarely.

The 7Q10 flow in this river is 313.3 cfs, so upstream volume Q_U, is set to 313.3. The background (upstream) fecal coliform concentration is 100 MPN/100ml, so C_U is set to 100. The

WQS stipulates that not more than 10 percent of samples taken during any 30-day period exceed 1,000 MPN/100 ml; thus receiving water concentration C_{RW} is set at 1000. Given 7Q10 flow in the receiving water, the mass balance equation may be rearranged to express the CSO concentration that just meets the standard, in terms of the CSO flow volume:

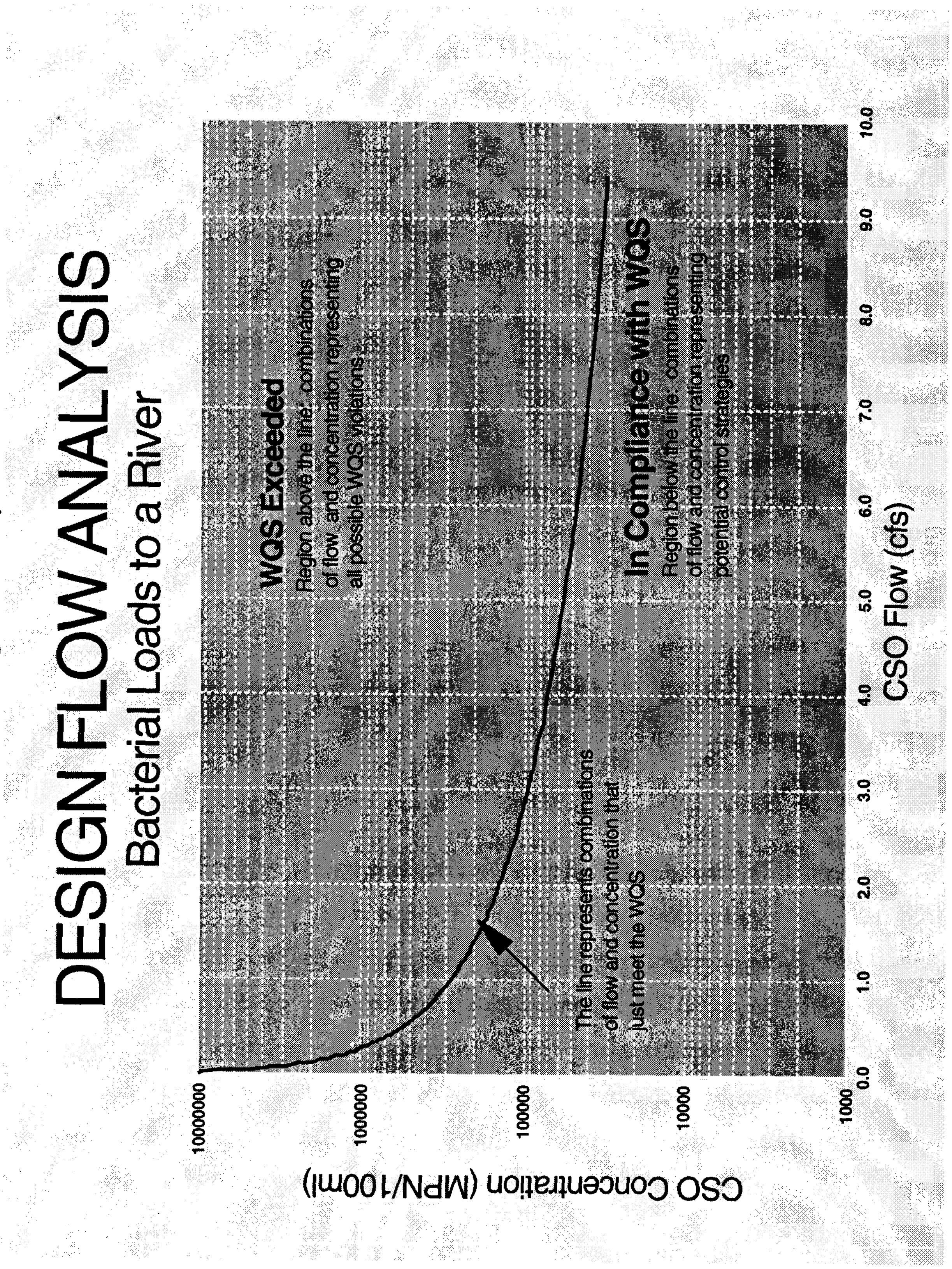
$$C_{CSO} = \frac{C_{RW}(Q_{CSO} + Q_U) - C_U Q_U}{Q_{CSO}} = \frac{1000(Q_{CSO} + 313.3) - 100 \times 313.3}{Q_{CSO}}$$

The equation treats both the concentration and flow from the CSO as variables, unlike a standard wasteload allocation for a point source, where flow is usually considered constant. For a given CSO concentration, the capacity of the receiving water increases as increased CSO volume provides additional dilution capacity. Therefore, the relationship between allowable concentration and CSO flow is not linear. The necessary levels of control on CSOs are not represented by a single point, but rather by a set of combinations of concentration and flow that meet the water quality criterion.

Exhibit 9-3 shows combinations of CSO concentration and CSO flow that just meet the WQS at 7Q10 flow. The region below the line represents potential control strategies. For instance, for CSO flows below 1 cfs, the WQS would be met at the design low flow of 313.3 cfs in the receiving water when the concentration in the CSO remained below 0.28×10^6 MPN/100 ml. At a CSO flow of 6 cfs, however, the concentration must be below 0.048×10^6 MPN/100 ml for WQS to be attained.

Since the typical concentration of fecal coliforms in CSOs is approximately 2×10^6 MPN/100 ml, demonstrating attainment of the water quality criterion via a design low flow analysis would be difficult.

Exhibit 9-3. Design Flow Analysis



A design low flow analysis is often conservative because CSOs typically occur when the receiving water is responding to precipitation and higher-than-normal dilution capability is available. Further, while CSOs may occur during design low flows, this will be much rarer than the occurrence of the low flows themselves. Therefore, the use of the design low flow protects to a more stringent level than indicated since dilution effects are likely to be greater. Dilution effects can be considerable in areas of multiple sources of storm water discharge. Design flow analysis is usually not sufficient in circumstances involving multiple storm water discharges, highly sensitive habitats, and river areas particularly prone to sediment deposition.

Design Flow Frequency Analysis

A design flow frequency analysis differs from design flow analysis in that it also considers the probability of exceeding WQS at a given flow. Although still simple, the design flow frequency approach better tailors the level of CSO control to the WQS. The major difference between CSOs and steady-state sources is that CSOs occur intermittently, providing no load on most days but large loads on an occasional basis.

Box 9-1. Average Number of CSOs per Month in Example

Jan	0.32
Feb	0.16
Mar	2.23
Apr	3.10
May	0.52
Jun	0.13
Jul	0.19
Aug	0.03
Sep	0.13
Oct	0.13
Nov	0.32
Dec	0.42

Over the 31 years, 238 CSO events occur, giving an average of 0.64 events per month. However, CSO events are unevenly distributed throughout the year: over 31 years, only one CSO has occurred in August but 96 have occurred in April. Box 9-1 shows the average numbers by month.

Since most CSOs occur in spring, the probability of a water quality criterion exceedance needs to be calculated on a month-by-month rather than annual average basis. Here, reducing the relatively high number of overflows in April should result in attainment of the criterion in other months.

Additional refinements can focus more specifically on eliminating only those CSO events predicted to exceed WQS at actual receiving water flow. Not all of the April events result in such excursions; many are very small. Further, the dilution capacity of the receiving water tends to be high during the spring. Therefore, the analysis can be refined by considering a design flow appropriate to the month in question and then counting only those CSO events predicted to result in excursions above WQS at this flow. The resulting table of predicted receiving water concentrations can be analyzed to determine the percentage reduction in CSO volume needed to meet the WQS.

The design flow frequency analysis can give results that are overly conservative, because the analysis assumes low flow at the same time that it imposes a low probability of exceeding the standard at that low flow. This approach, then, pays a price for its simplicity, by requiring highly conservative assumptions. A less restrictive analysis would need information on the probability distribution of receiving water flows likely to occur during CSO events.

Statistical Analysis

The next level considers not only design low flows, but the whole range of flows experienced during a month. Although CSOs are more likely when receiving water flow is high, CSO events do not always have increased dilution capacity available. Clearly, however, CSOs will experience at least the typical range of dilution capacities. Therefore, holding the probability of excursions to a specified low frequency entails analyzing the impacts of CSOs across the possible range of receiving water flows, and not only design low flows.

This example assumes that the permittee has a predictive model of CSO volumes and concentrations and adequate knowledge of the expected distribution of flows based on 20 or more years of daily gage data. In short, the permittee knows the loads and the range of available dilution capacity but not the frequency with which a particular load will correspond to a particular dilution capacity. A Monte Carlo simulation can readily address this type of problem, and is used with data on CSOs in April, since this is the month with the highest average number of CSOs and is the only month in which overflows occur more than 10 percent of the time, on average.[4]

Exhibit 9-4 summarizes the April receiving water flows in a flow-duration curve, which indicates the percent of time a given flow is exceeded. The distribution of flows is asymmetrical, with a few large outliers. An analysis of flow data indicates that daily flows typically are lognormally distributed. April's flows are lognormal with mean natural log of 7.09, which is ln (1,200 cfs)[5], and standard deviation of 0.46.

The 31 years of CSS data include 96 overflow events in April. In the Monte Carlo simulation these 96 events were matched with randomly selected receiving water flows from the April flow distribution, for a total of 342 "Aprils" of simulated data. The number of events in which the 1,000 MPN/100 ml standard would be exceeded was then calculated, and the count for the month tabulated.

Exhibit 9-5 shows the results. Of the 342 Aprils simulated, 122 had zero excursions of the standard attributable to the CSS. The maximum number of predicted excursions in any April was 17. The average number for the month was 2.45.

This analysis more closely approaches the actual pattern of water quality excursions caused by the CSS. The objective implied by the WQS is three or fewer excursions per month. In Exhibit 9-5, the right-hand axis gives the cumulative frequency of excursions, expressed on a

[4] The Monte Carlo approach describes statistically the components of the calculation procedure or model that are subject to uncertainty. The model (in this case, the simple dilution calculation) is run repeatedly, and each time the uncertain parameter, such as the receiving water flow, is randomly drawn from an appropriate statistical distribution. As more and more random trials are run, the resulting predictions build up an empirical approximation of the distribution of receiving water concentrations that would result if the CSO series were repeated over a very long series of natural flows. Monte Carlo analysis can often be performed using a spreadsheet. The resulting distribution can then be used for analyzing control strategies. Also see discussion in Section 8.3.

[5] For a lognormal distribution, the mean is equal to the natural log of the median of the data (7.09 = ln (median)). Therefore, the median April flow = $e^{7.09}$ = 1,200 cfs.

Exhibit 9-4. Flow Duration Curve

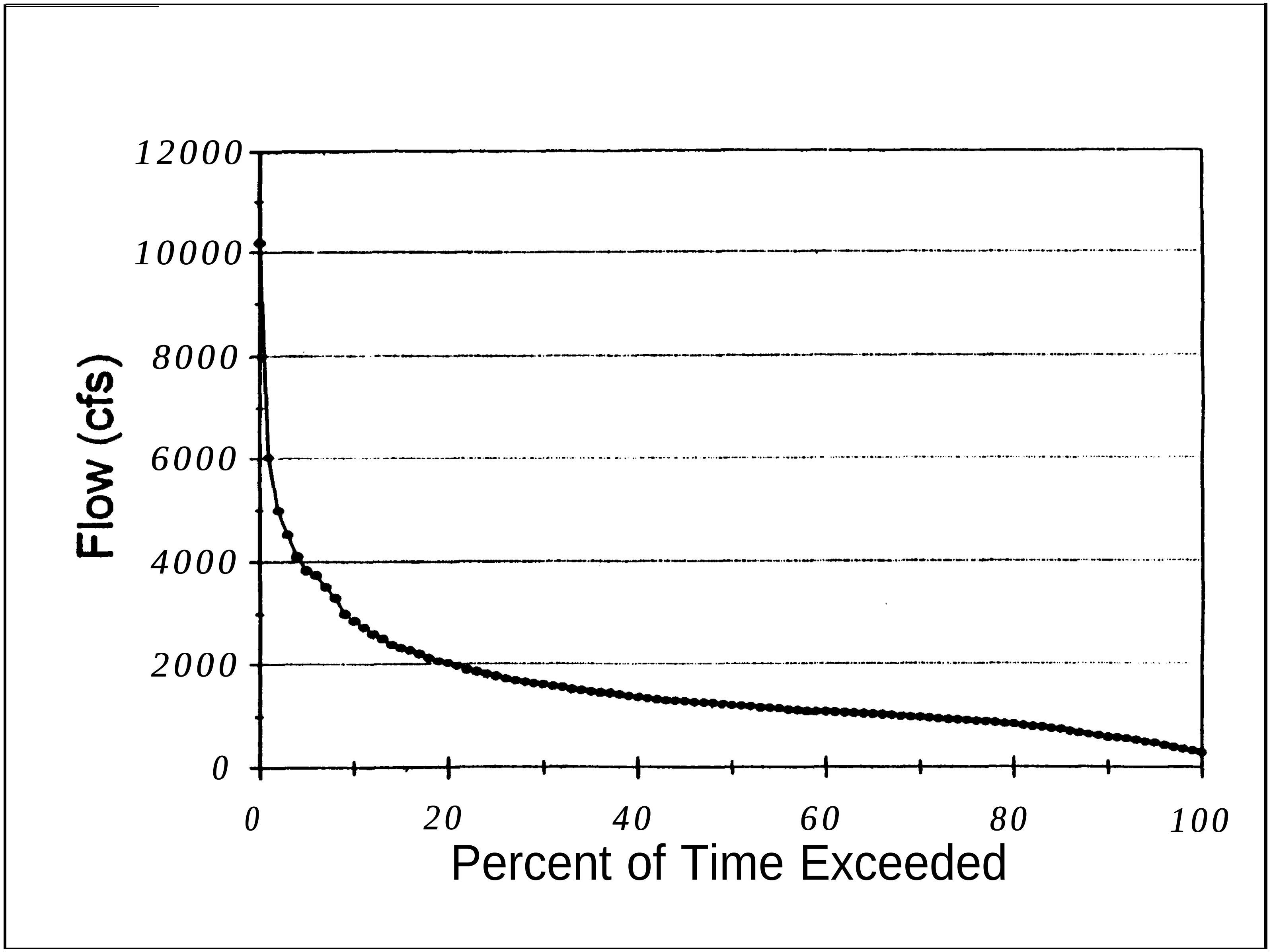
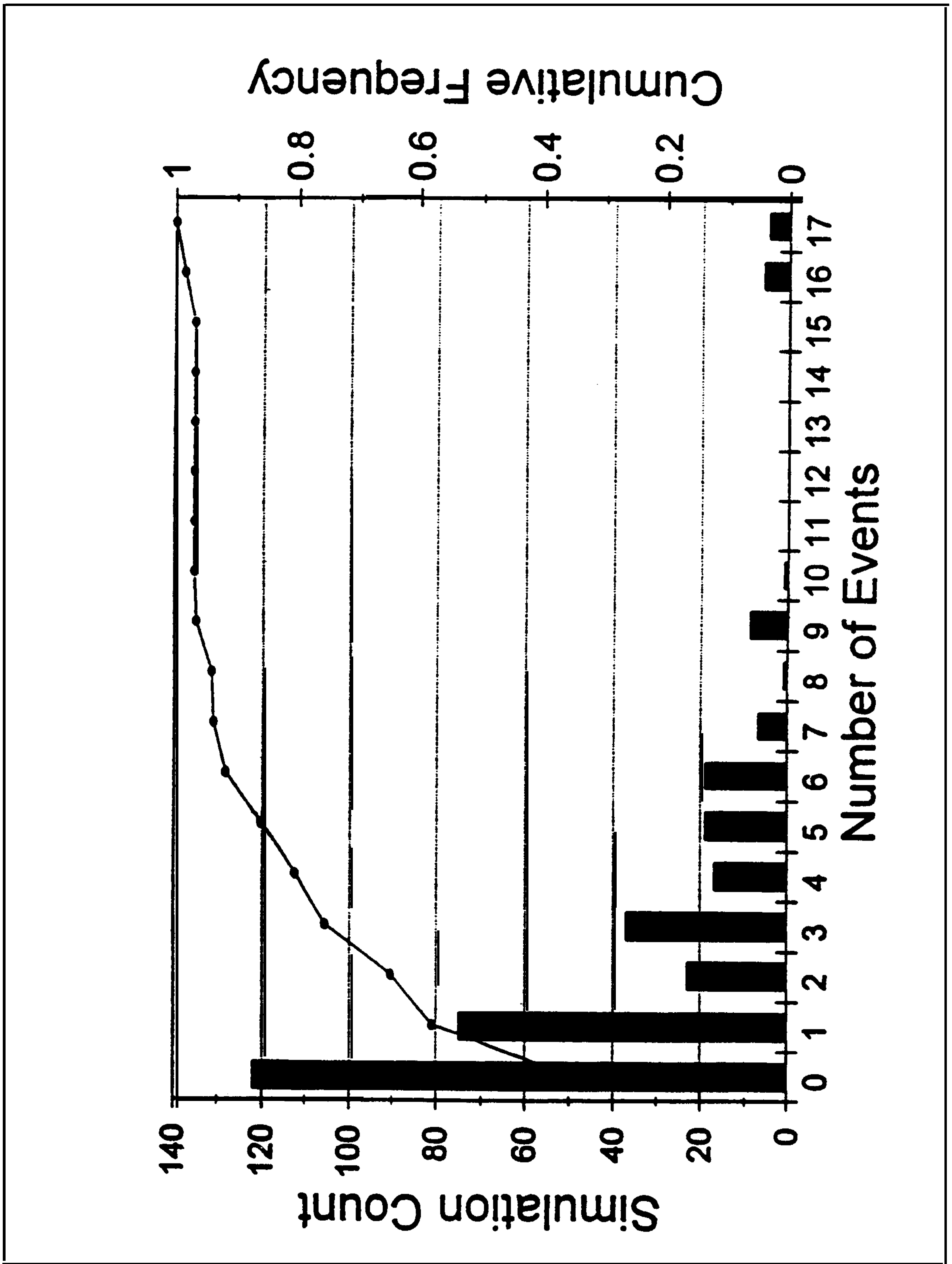


Exhibit 9-5. Expected Exceedances of Water Quality Criterion



zero-to-one scale. Of the 342 simulated Aprils, over 75 percent were predicted to have three or fewer excursions, leaving 25 percent predicted to have four or more. Note that the 11 simulated Aprils with either 16 or 17 excursions all result from the same month of CSS data, corresponding to an abnormally wet period.

Once set up, the Monte Carlo simulation readily evaluates potential control strategies. For instance, to evaluate a control strategy with the goal of a 20-percent reduction in CSO flow and a 30-percent reduction in coliform levels, the Monte Carlo simulation is rerun for these reduced CSO flows and coliform levels. The results show that of the 342 simulated Aprils, 82 percent were predicted to meet the water quality criterion. Although the Monte Carlo analysis introduces a realistic distribution of flows, it may still result in an overly conservative analysis for how CSOs correlate with receiving water flows, since it involves using a distribution, such as lognormal, which at best approximates the true distribution of flows.[6] A more exact analysis needs accurate information about the relationship between CSO flows and loads and receiving water dilution capacity.

Continuous Watershed Simulation

The most precise approach may be a dynamic simulation of both the CSS and the receiving water. This approach uses the same time series of precipitation to drive both the CSS/CSO model and the receiving water model. In cases where a dynamic simulation of the entire watershed would be prohibitively expensive, and where sufficient flow and precipitation records are available, the permittee may combine measured upstream flows and a simulation of local rainfall-runoff to represent the receiving water portion of the simulation.

As above, receiving water modeling entails an extremely simple dilution calculation. Determining the data for the dilution calculation by simulating dilution capacity or flows, and the

[6] An analysis of flow distribution must be made so that the appropriate Monte Carlo distribution and range are calculated.

analysis of the data, introduces complexity. This analysis uses a model that accurately predicts the available dilution capacity corresponding to each CSO event. Such a model accurately represents the actual coliform counts in the receiving water and enables the permittee to determine which events exceed the standard of 1,000 MPN/100 ml.

Exhibit 9-6 presents the results as the count of CSO events by month which result in receiving water concentrations greater than or equal to 1,000 MPN/100 ml. For 31 years of data, only three individual months are predicted to have more than three days (i.e., greater than 10 percent of the days in a month) in excess of the standard. Consequently, excursions above the monthly percentile goal occur only about 0.8 percent of the time. Further, the return period for years with exceedances of this standard is 10.3 years (3 occurrences over 31 years). Although the CSS produces relatively frequent overflows, the rate of actual WQS exceedances is quite low. Exhibit 9-7, which plots CSO volumes versus receiving water flow volume, illustrates why WQS exceedances remain rare. This figure shows that all the CSO events have occurred when the receiving water is at flow above 7Q10. Furthermore, most of the large CSO discharges are associated with receiving water flows well above low flow. Although this excess dilution capacity reduces the effect of the CSO pollutant loads, demonstrating compliance also necessitates careful documentation of the degree of correlation.

Of course, no simulation represents reality perfectly. Further, the model is based on precipitation series or rainfall-runoff relations that are likely to change with time. Therefore, an analysis of the uncertainty present in predictions should accompany any predictions based on continuous simulation modeling. An LTCP justified by the demonstration approach should include a margin of safety that reflects the degree of uncertainty in the modeling effort.

Exhibit 9-6. Excursions of Water Quality Criterion by Month

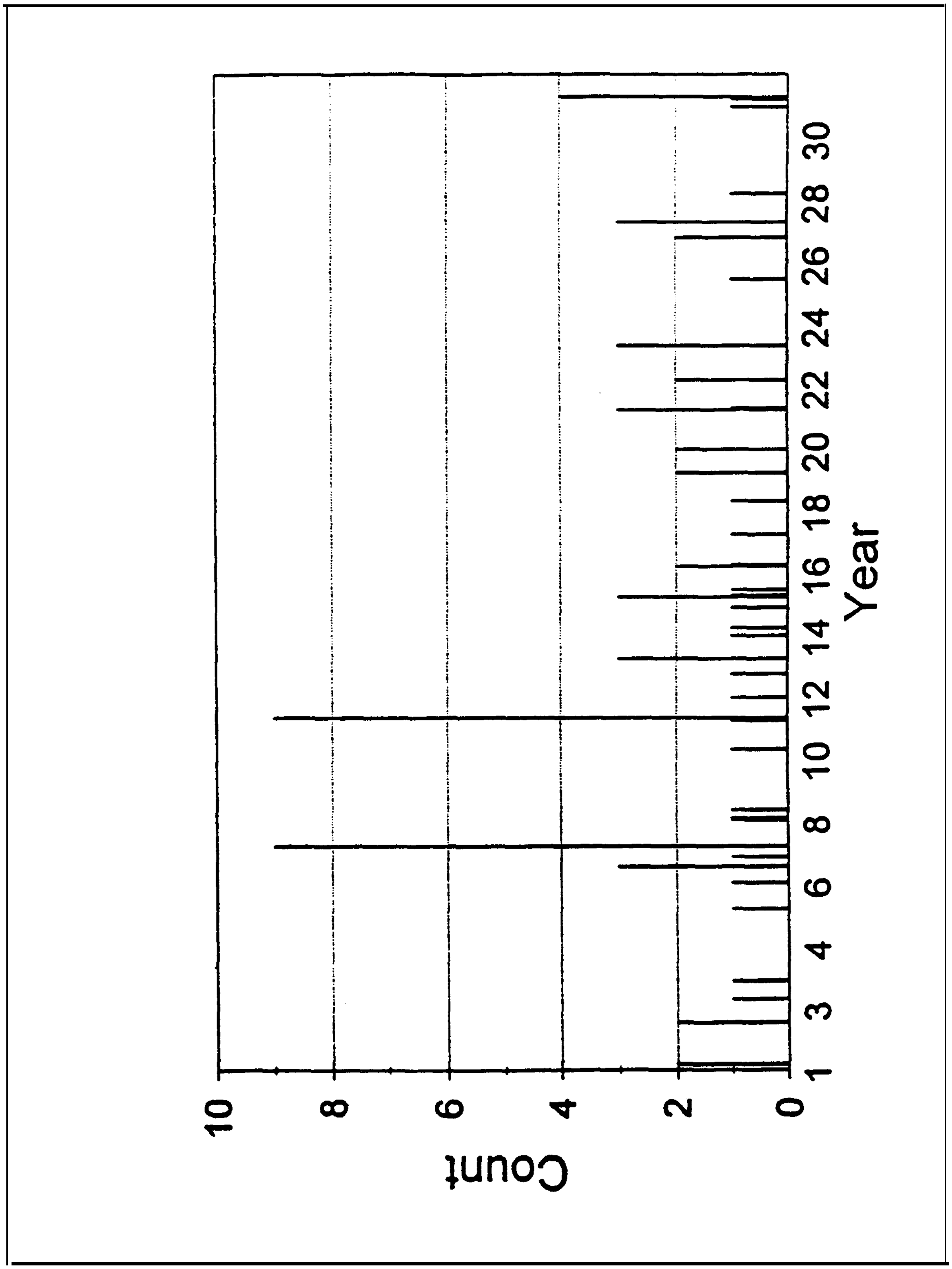
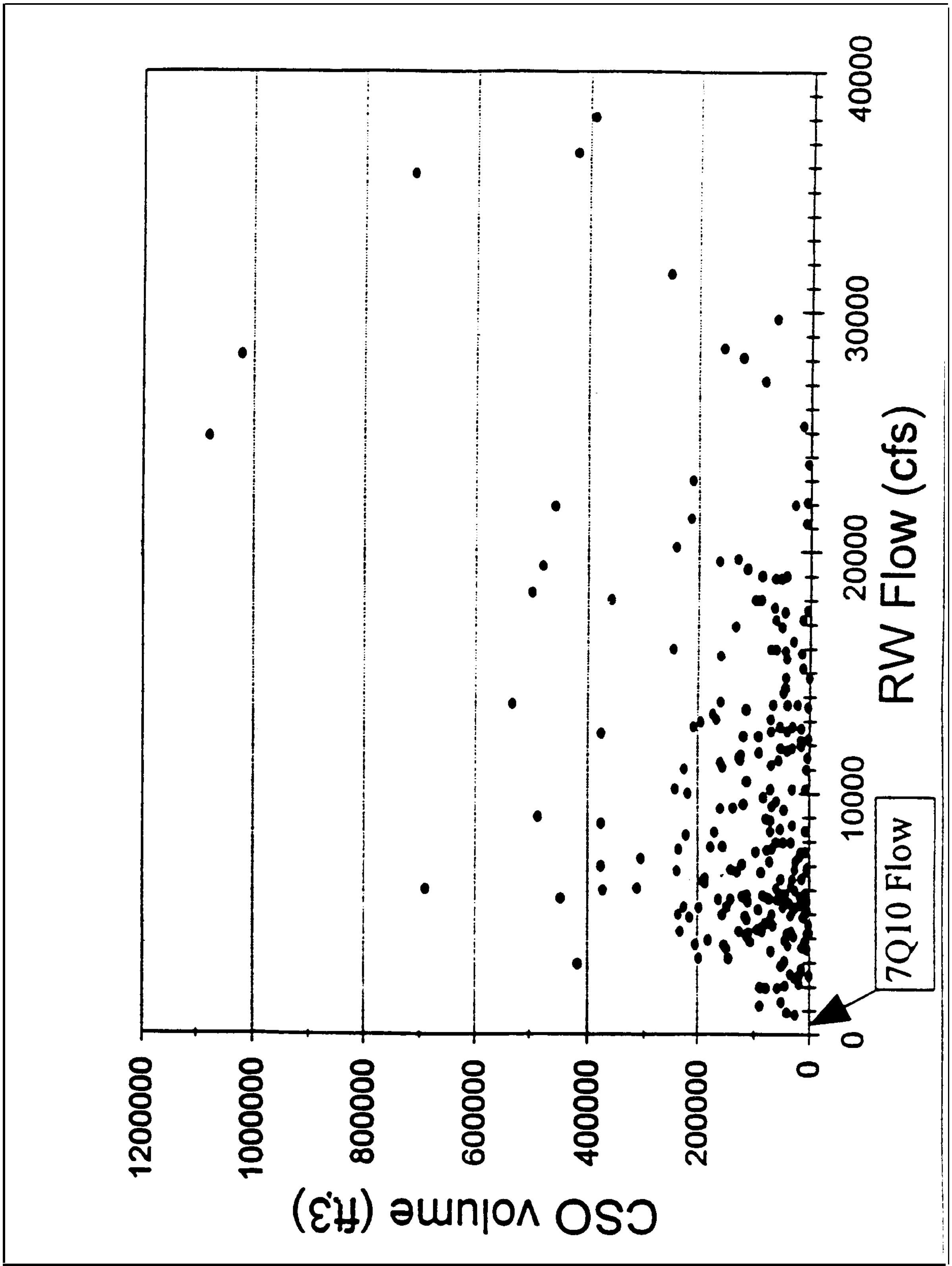


Exhibit 9-7. Receiving Water Flow During CSOs



9.3.2 Example 2: Bacterial Loads to an Estuary

The second example involves bacterial WQS in a tidal estuary. Like the previous example, it attempts to estimate the frequency of excursions of the WQS. However, the fate and transport of bacteria in an estuarine system is more complex than the transport in freshwater systems. Estuaries are both dispersive and advective in nature which creates considerable variations in the water quality. Dispersion is caused by the effects of tidal motion, which is the result of upstream and downstream currents. Advection is the result of the freshwater flow-through in the estuary. Exhibit 9-8 is a map of the estuary with the locations of the CSO outfall, mixing zone, and two sensitive areas (beach and shellfish bed) with more-restrictive bacterial standards.

As in the previous example, WQS for fecal coliform are expressed as a geometric mean of 400 MPN/100 ml and not more than 10 percent of samples in a 30-day period above 1,000 MPN/100 ml. The shell fishing and bathing areas have more restrictive WQS, specifying that the 30-day geometric mean of fecal coliform counts not exceed 200 MPN/100 ml on a minimum of five samples and that no more than 20 percent of samples exceed 400 MPN/100 ml.

Design Condition Analysis

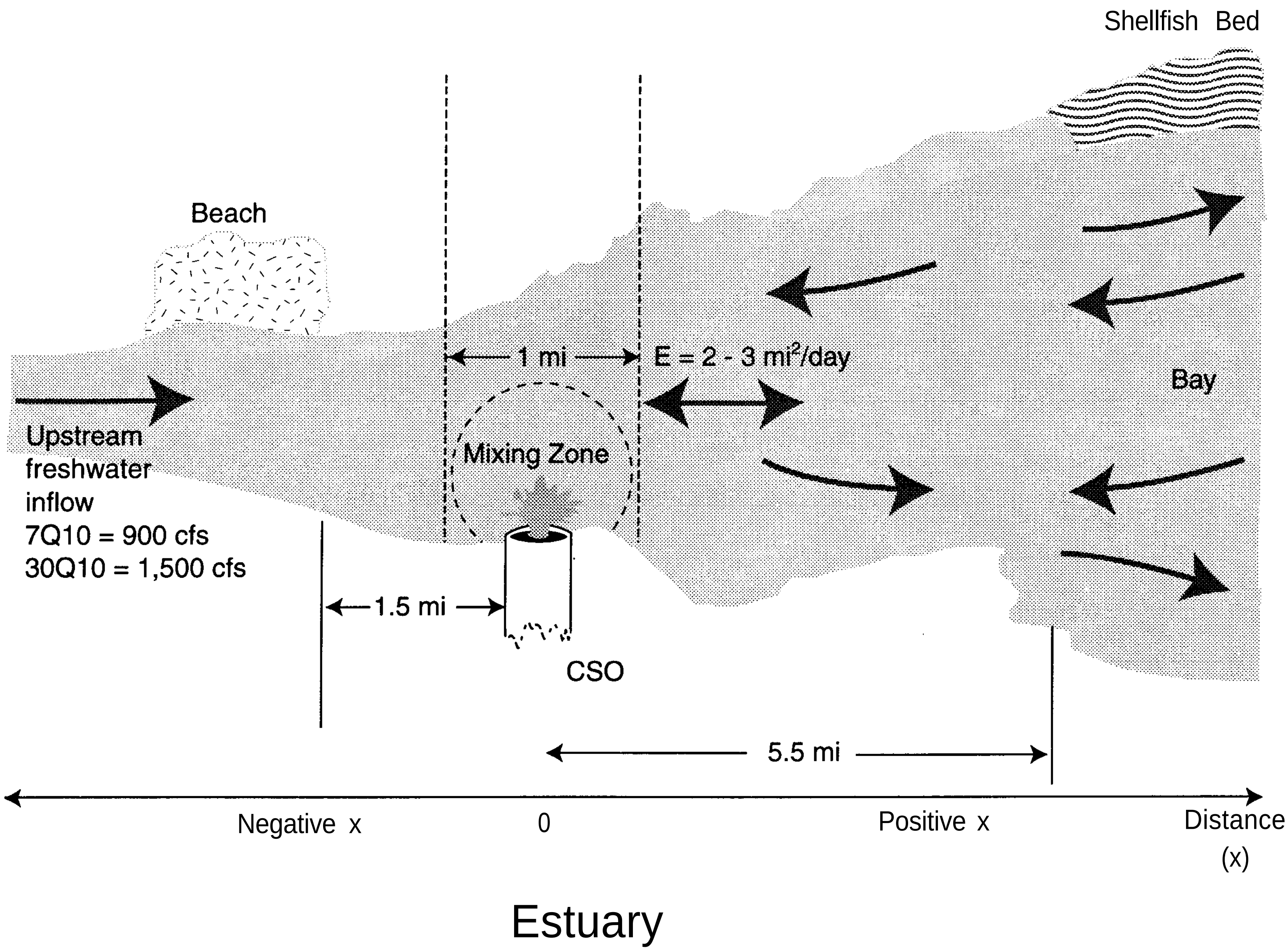
The use of a “design-condition” approach in an estuary requires the use of a model which includes several simplifications to the overall transport. The simplifications can be summarized through the following assumptions:

1. The estuary is one-dimensional. It is not strongly stratified near the source and the longitudinal gradient of bacterial concentration is dominant.

2. The bacterial concentration is described as a type of average condition over a number of tidal cycles. In other words, the model does not describe the variations in bacterial counts within the tidal cycle, but from one tidal cycle to the next.

3. The estuary is in a steady-state condition and area, flow, and reaction rate are constant with distance.

Exhibit 9-8. Map for Example 2



Under these assumptions, the following mass balance equation can be derived for an infinitely long estuary with a waste input at x = 0. This differential equation is often referred to as the one-dimensional advection-dispersion equation.

$$E\frac{d^2n}{dx^2} - U\frac{dn}{dx} - Kn = 0 \qquad (1)$$

for $n = n_0$ at $x = 0$ (2)

$n = 0$ at $x = +/- \infty$ (3)

where E is the tidal dispersion (mi^2/day), U = Q/A the net non-tidal velocity, K is the bacteria die-off rate (/day), and n is the bacterial concentration (MPN/100 ml).

The solutions to equation (1) with conditions (2) and (3) are:

$n = n_0 \exp(j_1 x)$ for $x \leq 0$

$n = n_0 \exp(j_2 x)$ for $x \geq 0$

where $j_1 = \frac{U}{2E}(1+\alpha)$ the coefficient j_1 is associated with negative values of x

$j_2 = \frac{U}{2E}(1-\alpha)$ the coefficient j_2 is associated with positive values of x

and $n_0 = \frac{W}{Q\alpha}$ n_0 is the concentration at x = 0, the point of the CSO input and W is the CSO input load to the estuary

where $\alpha = \sqrt{1+4KE/U^2}$ a is a coefficient that accounts for the dispersive nature of the estuary.

The ratio KE/U^2, referred to as the Estuary Number, strongly controls the character of the solution. As KE/U^2 approaches zero, advection predominates and the concentrations in the estuary become increasingly similar to the transport in a stream and, as KE/U^2 becomes large, the concentrations approach those in a purely dispersive system. Note that in a well-mixed river with no tides, a is equal to 1, and n_0 is given by the input CSO load divided by the flow. In an estuary, the concentration is reduced by the coefficient a due to the transport of the substance upstream and downstream because of tidal mixing.

Box 9-2. Assumptions for Estuarine CSO Example

Upstream Flows		
7Q10	=	900 cfs
U (7Q10)	=	1.5 mi/day
30Q10	=	1,500 cfs
U (30Q10)	=	2.5 mi/day
Estuary		
A	=	10,000 ft^2
E	=	2–3 mi^2/day
T	=	27°C
K	=	1.11/day
Unstratified		
CSO		
C	=	2×10^6 coliforms/100 ml
Q_e	=	0.1 MGD as maximum average per month, 2 MGD as daily maximum

Selected data for the example are presented in Box 9-2. A mixing zone of 0.5 mile up- and down-estuary is allowed. The beach location (1.5 miles up-estuary of the outfall) and the shellfish bed (5.5 miles down-estuary of the outfall) are of particular interest. The geometric mean requirement of the water quality criterion is taken as an average condition over time for scoping; that is, the 30-day time frame for this analysis is assumed sufficiently long to allow the variability in the load, as well tidal cycles, to be averaged out. The model was applied to a variety of conditions, including freshwater flow at 7Q10 and 30Q10 levels and bacteria loads at the estimated event maximum daily average load and expected maximum 30-day average load. Because the result depends on the value assigned to the dispersion coefficient, sensitivity of the answer to dispersion coefficients of 2 mi^2/day and 3 mi^2 /day, representing the expected range for the part of the estuary near the outfall, was examined.

Exhibit 9-9 displays the results of this analysis. It predicts fecal coliform counts at different locations in the estuary under different assumptions for tidal dispersion and non-tidal velocity.

Exhibit 9-9. Steady-State Predictions of Fecal Coliform Count (MPN/100 ml)

Upstream Flow	**7Q10: 900 cfs**		**30Q10: 1,500 cfs**			
Load	**Event Maximum Load**				**Average Load**	
Dispersion (mi²/day)	**E = 2**	**E = 3**	**E = 2**	**E = 3**	**E = 2**	**E = 3**
Upstream Mixing Zone (x = −0.5 mile)	1672	1640	1192	1302	60	66
Downstream Mixing Zone (x = 0.5 mile)	2,420	2,096	2,200	1,960	110	98
Beach (x = −1.5 mile)	504	666	246	414	12	20
Shellfish Bed (x = +5.5 mile)	238	268	378	386	18	18
Applicable WQS (MPN/100 ml) – shellfish/bathing areas – other	 400 1,000	 400 1,000	 400 1,000	 400 1,000	 200 400	 200 400

It is most appropriate to compare the geometric mean criteria to the 30Q10 upstream flow and average load (since the standard is written as a 30-day average), and the percentile standards to the 7Q10 upstream flow and event maximum load. Scoping indicates that the CSOs may cause the short-term criterion to be exceeded at the mixing zone boundaries and may cause impairment at the up-estuary beach. Increasing the estimate of the dispersion coefficient increases the estimated concentration at the beach, reflecting increased up-estuary “smearing” of the contaminant plume, which illustrates that the minimum mixing power may not be a reasonable design condition for evaluating maximum impacts at points away from the outfall. Potential WQS excursions at the beach are a concern only at low upstream flows, since the combination of average loads and 30Q10 freshwater flows is not predicted to cause impairment. In evaluating impacts at the beach, recall that scoping was conducted using a one-dimensional model, which averages a cross-section. If the average is correctly estimated, impacts at a specific point (e.g., the beach) may still differ from the average. Concentrations at the beach may be higher or lower than the cross-sectional average, depending on tidal circulation patterns.

The design condition analysis identifies instantaneous concentrations at the down-estuary boundary of the mixing zone and the beach as potential compliance problems. In this example, sensitivity analysis was performed on the dispersion coefficient, which varied within an expected range. Similar analysis can be made using other sensitive design variables such as temperature, which influences the coliform die-off rate and ultimately the predicted coliform count. Numerical experiments with the design condition scoping model suggest that a target 25-percent reduction in CSO flow volume would provide for the attainment of WQS.

Design Flow Frequency Analysis

The design condition analysis addresses the question of whether there is a potential for excursions of WQS. It does not address the *frequency* of excursions, which depends on (1) the frequency and magnitude of CSO events and (2) the dilution capacity of the receiving water body at the time of discharge. Note that, in the estuary, the range of dilution capacities (on a daily basis) is less extreme than in the river, because the tidal influence is always present, regardless of the level of upstream flows. To obtain an upper-bound (conservative) estimate of the frequency of excursions, an analysis of the monthly or seasonal frequency of CSO events should be combined with a design dilution capacity appropriate to that month.

Statistical Analysis

The design flow analyses of the previous two sections contain a number of conservative simplifying assumptions:

(1) They assume a steady (rather than intermittent) source

(2) They assume a design minimum dilution capability for the estuary

(3) They do not account for many of the real-world complexities of estuarine mixing

(4) They do not account for the effects of temperature and salinity on bacterial die-off.

The scoping analysis can be improved by considering a full distribution of probable upstream flows in a Monte Carlo simulation. The expected range of hydrodynamic dispersion coefficients could also be incorporated into the analysis.

Watershed Simulation

Building a realistic model of contaminant distribution and transport in estuaries is typically resource-intensive and demanding. A watershed simulation may, however, be needed to demonstrate compliance for some systems where the results of conservative design flow analyses are unclear. Detailed guidance on the selection and use of estuarine models is provided in EPA's *Wasteload Allocation* series, Book III (Ambrose et al, 1990; Martin et al., 1990).

9.3.3 Example 3: BOD Loads

The third example concerns BOD and depletion of DO, another important water quality concern for many CSSs. Unlike bacterial loads, BOD impacts are usually highest downstream of the discharge and occur some time after the discharge has occurred.

The CSS in an older industrial city has experienced frequent overflow events. The CSOs discharge to a moderate-sized river on a coastal plain. In the reach below the CSS discharge, the river's 7Q10 flow is 194 cfs, with a depth of 5 feet and a velocity of 0.17 ft/s. Above the city, velocities range from 0.2 to 0.3 ft/s at 7Q10 flow. A major industrial point source of BOD lies 18 miles upstream. A POTW with advanced secondary treatment discharges three miles upstream of the CSO (Box 9-3).

The river reach below the city has a designated use of supporting a warm water fishery. For this designation, State criteria for DO are a 30-day mean of 7.0 mg/l and a l-day minimum of 5.0 mg/l. The State also requires that WLAs for BOD be calculated on the basis of the l-day minimum DO standard calculated at 7Q10 flow and the maximum average monthly temperature. The 5.0 mg/l criterion is not expressed in a frequency-duration format; the 1-day minimum is a fixed value, but evaluation in terms of an extreme low flow of specified recurrence implicitly assigns an acceptable frequency of recurrence to DO 1-day average concentrations less than 5.0 mg/l. (The State criterion for DO is thus hydrologically-based and is roughly equivalent to maintaining an acceptable frequency of biologically-based excursions of the water quality criteria for ambient DO.)

Box 9-3. Assumptions for BOD Example

CSO Discharge (at maximum load)
- BOD_5 = 200 mg/l
- $CBODU/BOD_5$ = 2.0
- NBOD = 0 mg/l
- Q_e = 4 MGD

Point Source Effluent Upstream
- Distance Upstream = 18 mi
- BOD_5 = 93 mg/l
- $CBODU/BOD_5$ = 2.5
- NBOD = 0 mg/l
- Q_e = 5 MGD

POTW
- Distance Upstream = 3 mi
- BOD_5 = 11.5 mg/l
- Q_e = 10 MGD

Reaction Parameters
- T = 27°C
- $K_a = [12.9 \times U^{1/2}/H^{3/2}] \times (1.024)^{(T-20)}$
- where U = avg stream velocity (ft/s)
- and H = average depth (ft)
- $K_d = K_r = 0.3 \times (1.047)^{(T-20)}$
- SOD (below CSS) = 0.3 mg/l-day
- SOD (elsewhere) = 0

Upstream Background
- BODU = 1 mg/l
- DOD = l mg/l

Design Condition Analysis

A conservative assessment of impacts from the CSS can be established by combining a reasonable worst-case load (the maximum design storm with a 10-year recurrence interval) with extreme receiving water design conditions. Limited monitoring data and studies of other CSO problems suggested that a reasonable worst-case estimate was a l-day CSO volume of 4 MGD, with an average BOD_5 concentration of 200 mg/l.

As described in Chapter 8, initial scoping was carried out using a simple, steady-state DO model (see Section 8.3.1, Rivers-Oxygen Demand/Dissolved Oxygen subsection)[7]. The initial scoping assumes the presence of the upstream industrial point source and the POTW, and the estimated worst-case CSO load. All BOD_5 was initially assumed to be CBOD and fully available to the dissolved phase. Sediment oxygen demand (SOD), known to play a role in the reach below the CSS, was estimated at 0.3 mg/l-day. No SOD

[7] Similar DO analysis is discussed in Thomann and Mueller (1987).

was assumed for other reaches upstream of the CSO. This is a simplifying assumption that is sufficient for the scoping analysis described here. SOD in the river reach below the CSO has been included in the analysis since this is the reach of concern. Since there are many sources of SOD other than CSOs, contributions of SOD from other sources should be considered at the next level of analysis.

Results of the scoping model application are shown in Exhibit 9-10, which shows the interaction of the point source, POTW, and CSO. The exhibit combines two worst-case conditions: high flow from the episodic source and low (7410) flow in the receiving water. Under these conditions, the maximum DO deficit is expected to occur 7.5 miles downstream of the CSO, with predicted DO concentrations as low as 3.9 mg/l. Under such conditions, the CSO flow is approximately 25 percent of total flow in the river.

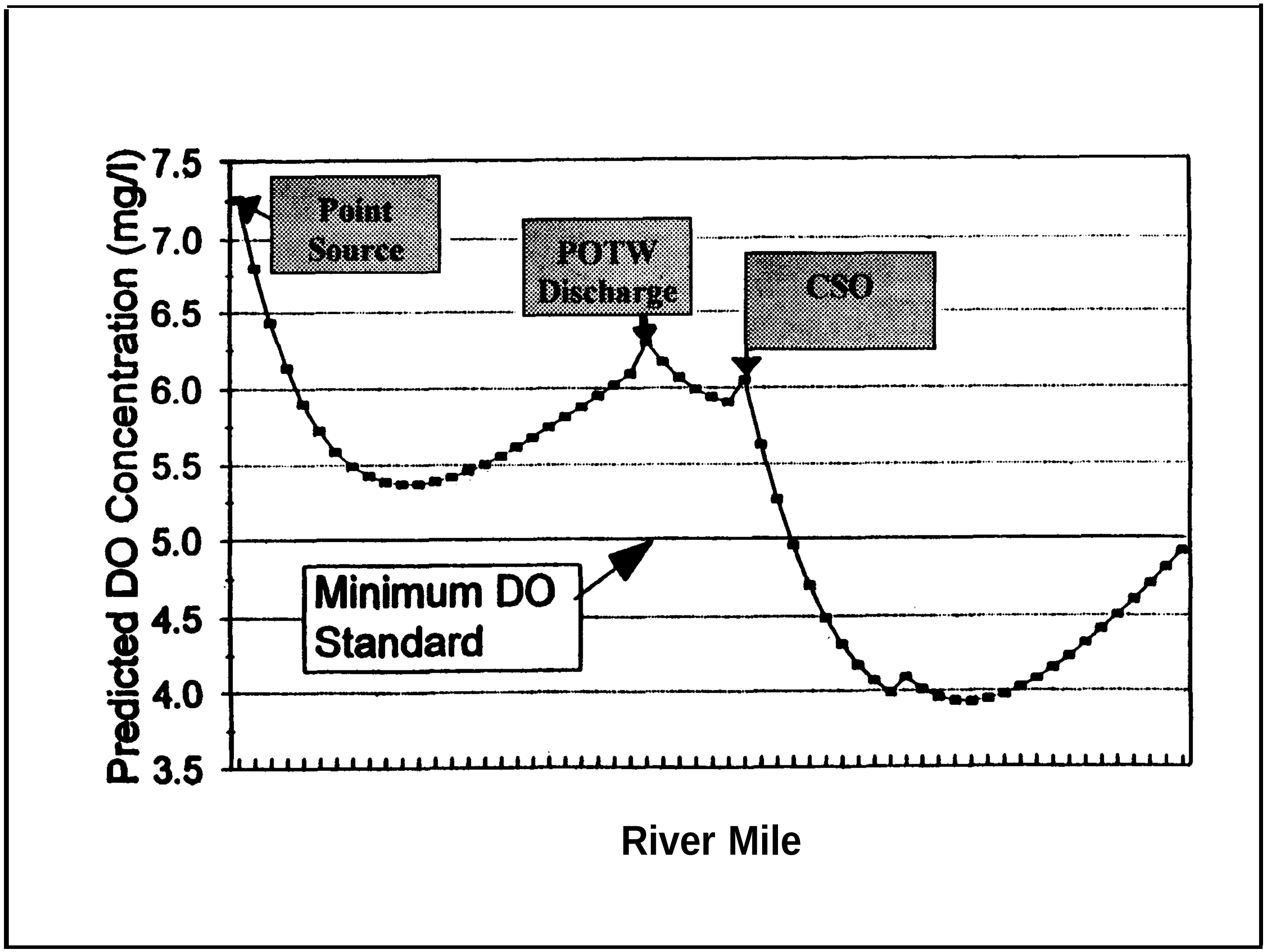
Design Flow Frequency Analysis

The State criterion called for a one-day minimum DO concentration of 5 mg/l, calculated at design low flow conditions for steady sources. Use of the 7Q10 design flow was interpreted as implying that an approximately once-in-three-year excursion of the standard, on average, was acceptable (U.S. EPA, 1991a).[8] As in the previous examples, the rate of occurrence of CSOs provides an upper bound on the frequency of WQS excursions attributable to CSOs. In this case, however, the once-in-three-year excursion frequency cannot be attained through CSO control alone. Instead, the co-occurrence of CSOs and receiving water flows must be examined.

To accommodate this relationship, the design flow model can be modified to assess the dependence of DO concentrations on upstream flow during maximum likely loading from the CSO. Design flow was simulated using the worst-case CSO flow over a variety of concurrent upstream

[8] The average frequency of excursions is intended to provide an average period of time during which aquatic communities recover from the effects of the excursion and function normally before another excursion. Based on case studies, a three-year return interval was determined to be appropriate. The three-year return interval was linked to the 7Q10 flow since this flow is generally used as a critical low flow condition.

Exhibit 9-10. Design Condition Prediction of DO Sag



flows, since upstream flows affect both the dilution capacity of the river and the velocity of flow and reaeration rate. As shown in Exhibit 9-11, the estimated DO concentrations depend strongly on upstream flow. Note that WQS are predicted to be attained if the upstream flow is greater than about 510 cfs. A flow less than 510 cfs occurs about five times per year, on average, in this segment of the river.

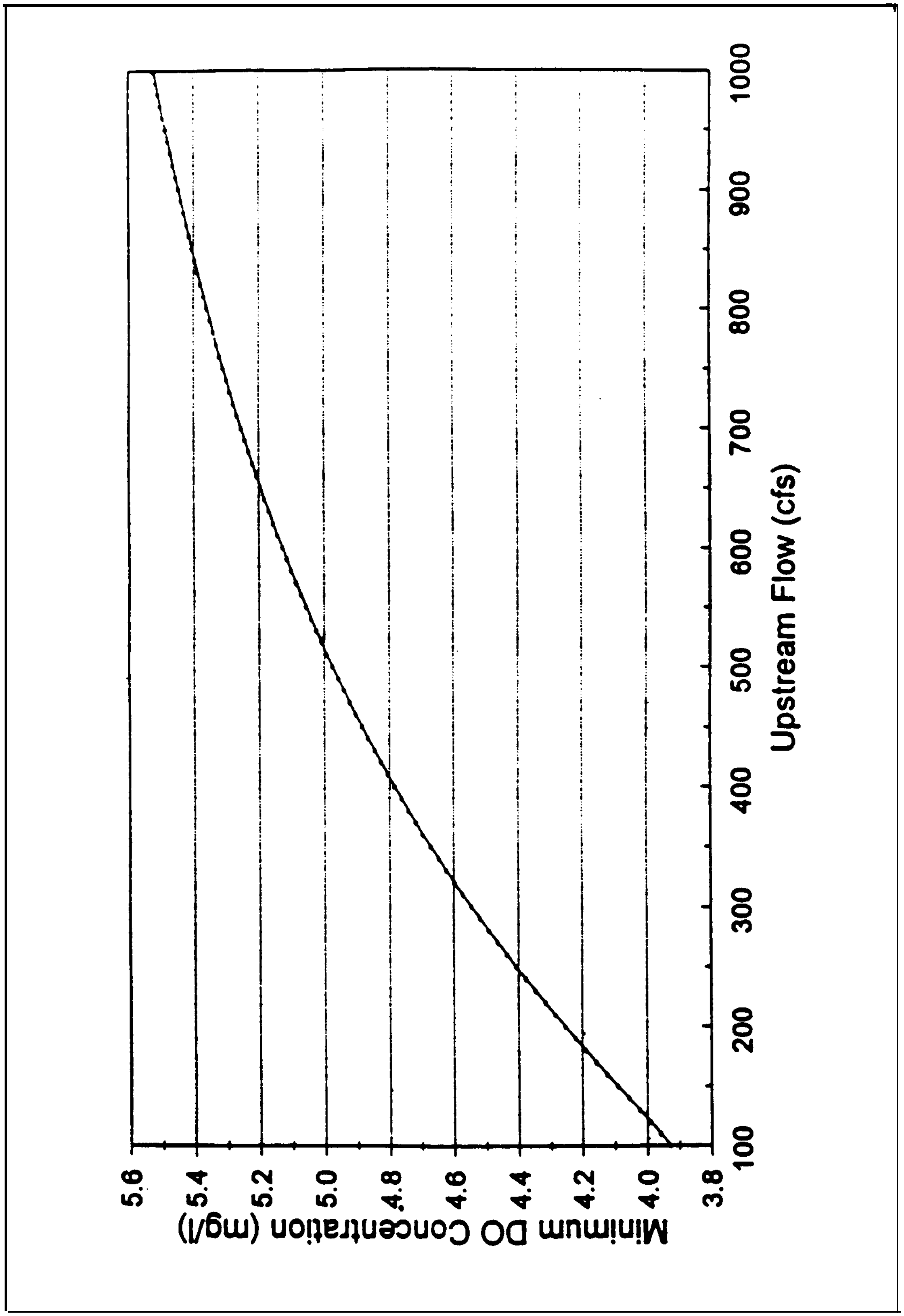
The target rate of WQS excursions is one in three years. An upper bound for the actual long-term average rate of excursions can be established as the probability that flow is less than 510 cfs in the river multiplied by the probability that a CSO occurs:

$$P_{exc} = p(Q<510cfs)f_{CSO}$$

where P_{exc} is the probability of a WQS excursion on any given day and f_{cso} is the fraction of days in the year on which CSO discharges occur, on average. Since the goal for excursions is once every three years, P_{exc} is set at 1/(3 x 365), or .000913. Since a flow less than 510 cfs occurs five times per year, p(Q<510) is 5/365, or .0137. Substituting these values into the equation yields f_{cso} = .000913/.0137 = 0.067. This implies that up to 24 CSOs per year will meet the long-term average goal for DO WQS excursions, even under the highly conservative assumption that all CSOs provide the reasonable maximum BOD load.

An important caveat, however, is that no other significant wet weather sources are assumed to be present in the river. In most real rivers, major precipitation events also produce BOD loads from storm water, agriculture, etc. Where such loads are present, conservative assumptions regarding these additional sources need to be incorporated into the scoping level frequency analysis.

Exhibit 9-11. Relationship Between DO Concentration and Upstream Flow



As with the other examples, further refinement in the analysis can be attained by examining the statistical behavior of the CSO and receiving water flows in more detail. For example, the use of a constant CSO load is a conservative, simplifying assumption that is appropriate for the scoping level analysis presented here. Dynamic continuous simulation models could be used to provide a more realistic estimate of the actual time series of DO concentrations resulting from CSOs.

9.4 SUMMARY

As illustrated in the preceding examples, no one method is appropriate for a particular CSS or for all CSSs, and a complex dynamic simulation is not always necessary. The method should be appropriate for the receiving water problem. The municipality (in cooperation with the NPDES authority) needs to balance effort spent in analysis with the level of accuracy required. However, as the first example illustrated, as additional effort is invested assumptions can usually be refined to better reflect the actual situation.